DRAFT

Part 1: Viability Criteria and Status Assessment of Oregon Coastal Coho

IMST and Stakeholder Team Draft

State of Oregon[1]

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I. Introduction

Populations of coho salmon (*Oncorhynchus kisutch*) that occur in coastal watersheds between Cape Blanco and the mouth of the Columbia River are being evaluated by NOAA Fisheries for listing under the Endangered Species Act (ESA). These populations, which have been designated a single Evolutionary Significant Unit (ESU) (Weitkamp et al. 1995), have been the focus of a considerable conservation effort by the State of Oregon and federal management partners. Much of this conservation effort has been developed and implemented under a planning framework called the Oregon Plan for Salmon and Watersheds (Oregon Plan). The Oregon Plan brings together various state governmental and non-governmental entities to implement conservation strategies for fish populations throughout Oregon, including those belonging to the Oregon Coastal Coho ESU. In this context, the Oregon Plan refers to the broad suite of conservation efforts implemented to improve the status of coho and their watersheds (e.g., harvest, hatcheries, habitat, etc)

The State of Oregon, in partnership with the National Marine Fisheries Service (NOAA Fisheries), initiated a collaborative project to address the conservation of coastal coho on the Oregon coast. The primary objectives of the Coastal Coho Project are to:
1. Assess Oregon Plan efforts to conserve and rebuild coastal coho populations.
2. Use the assessment to inform NOAA Fisheries' status review listing determination.
3. Use the assessment as a basis to seek legal assurances for local participants.
4. Use the assessment as a foundation for developing a recovery plan for coho.

This report addresses objective (2) of the Coastal Coho Project by providing a biological analysis of coho status relative to viability criteria (Part 1) and an assessment of conservation efforts to address the factors for decline and threats associated with the coastal coho ESU (Part 2)[2]. The two parts of this report address the requirements of the federal Policy for Evaluation of Conservation Efforts When Making Listing Decisions (PECE) used to inform ESA listing decisions (68FR15100).

There are five primary components of the draft biological analysis, which also serve as the primary organizational structure of this report:

1. Determination of the ESU, strata and population structure;

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[2] Part 2 of this report is currently under internal review. A draft will be submitted for IMST and Stakeholder Team review by mid January 2005.

2. Description of attributes used to define viability and assess fish status;
3. Development of specific criteria for each attribute used to define population viability;
4. Development of specific criteria for strata and ESU viability based on roll-up of population criteria; and,
5. Assessment of current status of coastal coho relative to population, strata and ESU viability criteria based on the key attributes described in (2).

II – Population and ESU Structure

Population and ESU Structure
The conceptual foundation for the biological criteria drew heavily from conservation principles for salmon and steelhead presented by (McElhany et al. 2000) in their publication entitled "Viable Salmonid Populations (VSP)". As such, individual populations were the primary units analyzed in assessing the conservation status of the ESU. Development of population-level biological criteria was therefore necessary to perform these population evaluations.   The status of the entire ESU was a product of these individual population assessments, rolled up to the ESU level and expressed in terms of the distribution and number of 'viable' populations d across the ESU. The identification of populations and their boundaries is illustrated in Figure 1 and corresponds with the draft populations proposed by the Oregon Working Group of the Coastal Coho Technical Recovery Team (OTRT) (Lawson et al. 2004).

The assumed population structure of the ESU was an integral part of how individual population assessments were condensed into a single status conclusion for the ESU. Oregon coastal coho are split into 67 populations. Most of these populations are classified as dependent, meaning they likely have always been small to persist for long periods (i.e., 100 years) without substantial reproductive support provided by strays from larger and more stable adjacent populations (Lawson et al. 2004). The remaining populations are larger and, at least historically, were capable of persistence over the long-term. These populations are referred to as independent populations. However, within this classification a further distinction is made between a "functionally independent" population and a "potentially independent" population. A "functionally independent" population is one that is so dominating (generally in large basins) that their population demographics are acting in manner that is functionally independent from all other populations. A "potentially independent" population is one that if isolated would be expected to persist on its own, however because of its location next to one or more large functionally independent populations, likely receives too many strays to be truly functioning in a demographically independent manner. Functionally independent and potentially independent populations are critical to survival of the ESU and provide the basis for this viability assessment... Lawson et al. (2004) identify 19 independent populations.
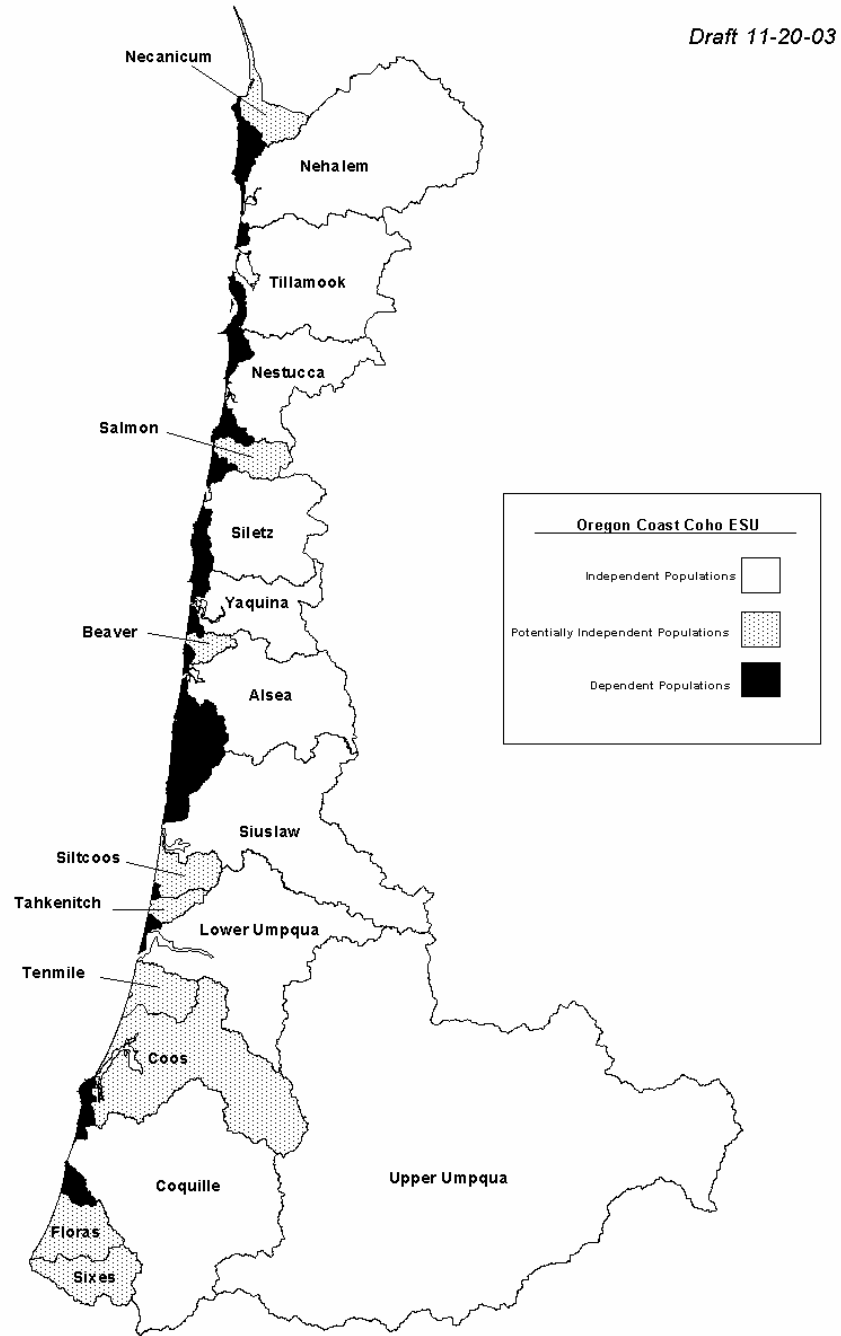
Figure 1 – Constituent populations of the coastal coho Evolutionary Significant Unit

Oregon coastal coho populations have also been clustered in 5 different groups called strata as presented in Table 1. These strata are intended to represent clusters of populations that share ecological or geographic similarities. Healthy strata are important in maintaining genetic and life history diversity in Oregon coastal coho.

Table 1. Delineation of geographic strata and constituent populations for Oregon coastal coho ESU.

Geographic strata	Populations	Independence category
Northern Stratum	Necanicum	Potentially Independent
	Nehalem	Functionally Independent
	Tillamook	Functionally Independent
	Nestucca	Functionally Independent
North-Central Stratum	Salmon	Potentially Independent
	Siletz	Functionally Independent
	Yaquina	Functionally Independent
	Beaver	Potentially Independent
	Alsea	Functionally Independent
	Siuslaw	Functionally Independent
Umpqua Stratum	Lower Umpqua	Functionally Independent
	Upper Umpqua	Functionally Independent
Lakes Stratum	Siltcoos	Potentially Independent
	Tahkenitch	Potentially Independent
	Tenmile	Potentially Independent
South-Central Stratum	Coos	Potentially Independent
	Coquille	Functionally Independent
	Floras	Potentially Independent
	Sixes	Potentially Independent

III – Population Attributes of Fish Performance

A key aspect from the VSP document incorporated in developing the biological criteria was the list of primary attributes that should be considered in determining whether or not a population is viable. These attributes are:

1. Abundance – the number of naturally-produced spawners.
2. Productivity – the life cycle survival commonly expressed as the number of recruits (progeny) produced per spawner (parent)
3. Distribution – the distribution of spawners among habitats within a population's home range
4. Diversity – indices of genetic variability related to a population's ability to adequately respond to unpredictable natural variations in the environment and retain those adaptive genetic characteristics that promote optimum survival in basin specific habitats.

The Native Fish Conservation Policy (NFCP) adopted by the Oregon Fish and Wildlife Commission (OAR 635-07-0502 through 0509) includes these same attributes of population viability plus the attribute of "persistence". Persistence is a forecast of future population health, stated in units of the probability of extirpation. Because a forecast of extirpation risk is critical to understanding the present condition of a population, biological criteria were developed for coastal coho to address this attribute.

>   5. Persistence – the forecast likelihood that the population will become extirpated in the future must be very low,

The OTRT has also included a persistence attribute in their draft viability criteria for coastal coho. In addition, ODFW's recovery plan for lower Columbia River coho contains biological criteria for the persistence attribute.

IV - Considerations for Criteria Development

Criteria Development Objectives

Success in the development of biological criteria for coastal coho was framed in terms of addressing four critical considerations: 1) the criteria must be scientifically defensible and tied to the most recent thinking on salmonid viability; 2) the criteria must be stated in units that can be unambiguously measured and utilize data that are readily available; 3) the criteria must be relatively easy to understand, and; 4) the application of the criteria to the status of coastal coho must yield consistent results that can be independently confirmed by others.

Fish Performance Based

Population viability criteria were developed for each attribute of fish performance. The measures of fish performance incorporated the collective effects of natural and anthropogenic factors, such as ocean conditions, harvest, hatcheries, habitat quality and quantity, predators, etc.

Another underlying concept used in development of the biological criteria for coastal coho was to rely on metrics related to fish performance rather than a more indirect approach of relying on 'habitat-based' metrics. Therefore, the population attributes proposed for use in this evaluation exclude the use of measured characteristics of a population's habitat. This does not imply that habitat issues are inconsequential for coastal coho. It is merely a reflection of a viewpoint that biological criteria need to be related to the performance of the fish. The impact of degraded habitat should be reflected in population metrics, such as abundance, productivity, and distribution.

For some species, population data are unavailable and habitat-based criteria must be used as a surrogate for the population performance indicators. In these cases inferences are drawn between habitat and a population trait such as abundance or productivity and then expanded across the range of the species. A benefit of habitat-based criteria is that the raw data necessary to perform the evaluation are frequently available, and if they are not available can be readily obtained. In contrast, population data spanning a sufficient

length of time and proportion of an ESU are rarely available. However, in the case of coastal coho in Oregon this rare condition exists. Abundance, productivity, and distribution data are available for nearly all populations within the ESU, from 1990 to present. Longer data sets from spawner index sites extend back to 1950. Therefore, given the availability of these data it was logical to craft 'fish-based' metrics for the biological criteria.

Again it is emphasized that taking this approach does not mean habitat related issues are ignored. Rather, habitat is treated as a primary factor that needs to be considered in explaining the performance of the various populations belonging to this ESU. Habitat, and similarly fisheries and hatcheries, are treated as potential causes for the condition of the population. Conservation strategies have been developed to improve habitat as a means to help rebuild coastal coho. However, the measure of whether these strategies have been successful is the response of the fish to these improvements, not the habitat improvements themselves. It is also recognized that in many cases corrective measures directed at habitat problems take a longer time to bear fruit and therefore monitoring of fish performance of a corresponding duration is necessary to demonstrate their effectiveness. In this context, habitat improvements to address limiting factors rarely receive immediate credit through fish performance metrics.

Effects of Marine Survival

Survival conditions for coho in the marine phase of their life history can cause wide fluctuations in subsequent adult returns and spawner abundance (Nickelson 1986). Smolt to adult survival rates between a high year and a low year are typically in the range of ten-fold. For example, if a basin consistently produces 100,000 coho smolts each year, the return of adults when ocean conditions are good may be 10,000 fish, whereas when ocean conditions are bad the same smolt number may produce a return of only 1,000 fish. This order of magnitude difference, which should be expected for a normally functioning natural coho population, complicates the development of biological criteria for coho. This effect is magnified by the basic life history of coho. Coho, unlike most other salmon and steelhead have a greatly simplified age structure, with about 80-90% of the fish returning to spawn as 3-year olds and the remaining 10-20% returning as 2-year olds, the latter almost entirely being males (jacks). Therefore, the effect of marine survival conditions in any particular year on the subsequent return is dramatic because nearly all of smolts that mature and return to spawn will do so in the same year. In contrast, for a species like steelhead the effect of variable marine conditions is dampened by the fact that the return of individuals from a particular smolt out-migration typically is divided among at least 3 different years. Therefore, large annual swings in abundance that would be cause for alarm with respect to steelhead populations should be less worrisome with respect to coho populations. For coho, large swings in abundance should be expected as a normal dynamic of the species. Therefore, coho have both an inherent higher risk of cohort collapse than other species, but they also have a greater capacity to quickly bounce back when survival conditions improve.

In addition to the wide range in survival rates for coho, these rates are known to fluctuate annually with variation in ocean temperature and upwelling (Nickelson et al. 1986;

Logerwell et al. 2003) as well as decadally in an apparent cyclic pattern associated with climate, as typified by the survival rate pattern illustrated in Figure 2 (Beamish and Bouillon 1993; Beamish et al. 2000). Because the low survival periods appear to correspond with climatic periods of warmer ocean temperatures, it has been speculated that global warming may adversely impact the long-term marine survival of Oregon coho. It is not clear whether the variations in marine survival are cycling about a steady mean, or if they are cycling about a declining trend-line as proposed by Lawson (1993). Regardless, this uncertainty makes the forecast of extirpation risk for coastal coho over the long-term conditional upon the assumption for marine survival. Because the rate and magnitude of a possible declining trend is unknown, it makes the task of assigning risk all the more difficult.

## Density Dependent Recruitment – Resilience at Low Spawner Abundance

The relationship between the number of spawners and the number of recruits they produce is not linear. A decrease in spawners by a value of 'x' does not yield an equal decrease in the number of adult recruits produced. Stated another way the ratio of recruits to spawners is not a constant. When the number of spawners is high, the smolt capacity of habitat is saturated and the number of recruits produced per spawner will be low. Conversely, when the number of spawners is low, there is very little competition among rearing juveniles for food and space and therefore a higher percentage of them survive to become smolts. Under these conditions the number of recruits produced per spawner will be high. Such expectations are based on considerable evidence in coho that the recruits per spawner metric is density dependent (Beidler et al. 1980; Nickelson and Lawson 1998; Nickelson 2003). For example, data for Tenmile Lake coho, graphically presented Figure 3 demonstrates this density dependent relationship. Further, the likelihood that much of this density dependent response occurs during the early portion of the freshwater rearing cycle is further supported by (Nickelson and Lawson 1998) and is summarized in Figure 4.

For coastal coho populations, the importance of this strong density dependent relationship is threefold. First, the recruit per spawner (R/S) ratio is not a constant and should not be treated as such; it is highly variable depending on spawner density. Second, spawner-recruit models that rely on the assumption of a single R/S ratio, such as the hockey stick are poorly suited for forecasting the recruitment response of coho populations when the density of spawners is low. Finally, as spawner abundance trends downward the resistance to further population decline becomes greater and greater because of the compensatory increase in the R/S ratio.

An effect of this strong density-dependent relationship is that even a relatively low number of spawners can produce enough smolts that, if these smolts experience good marine survival, the number of returning adults will rebound considerably. This pattern was observed throughout the Oregon coast ESU in the last recruitment cycle (1997 to 2002), especially for those populations in the northern portion of the range.
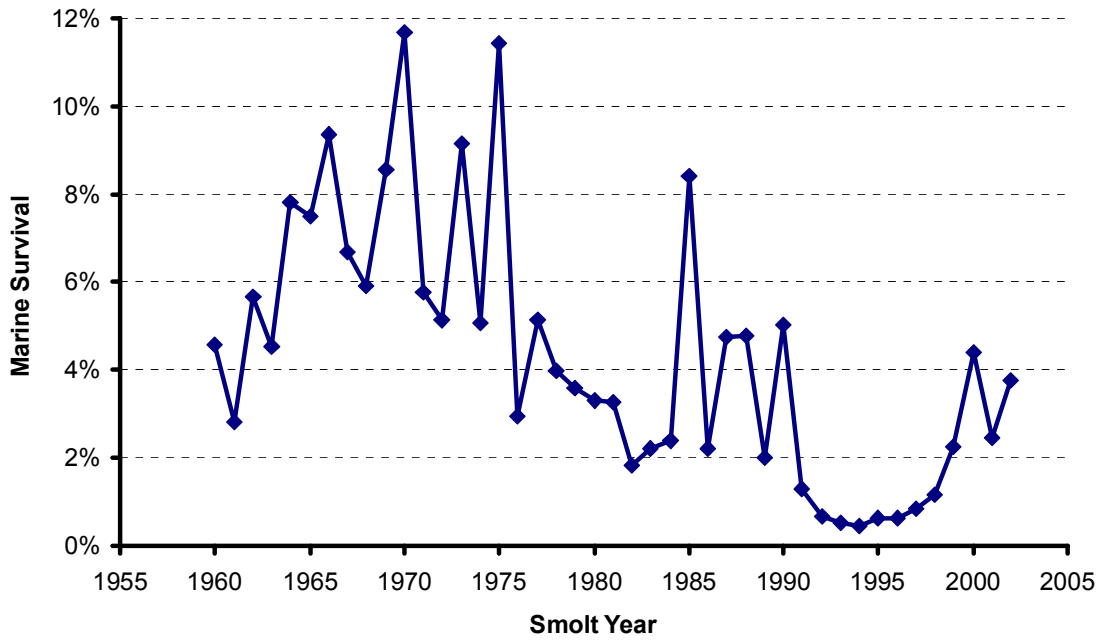
2a)
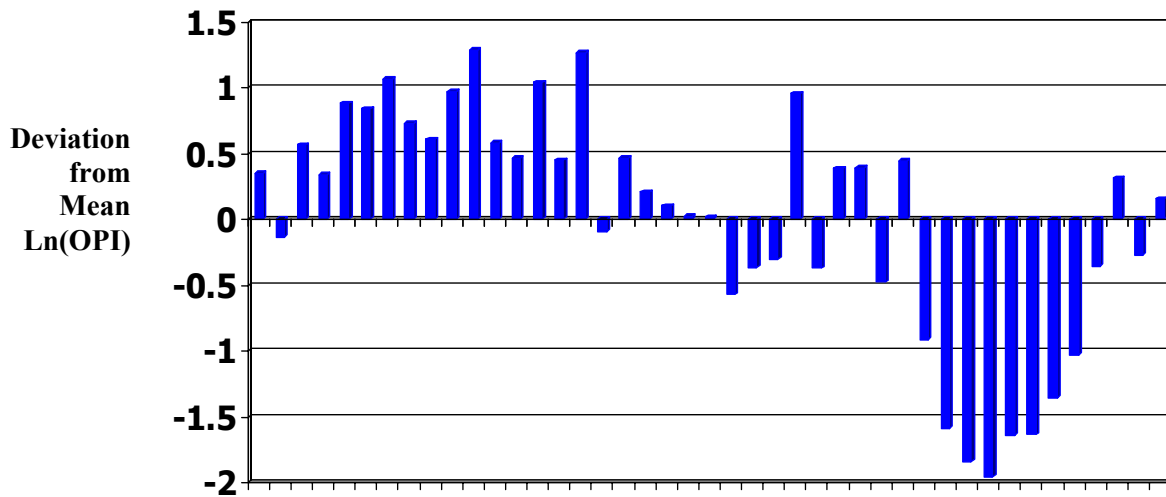
2b)

Figure 2. Estimated survival of Oregon Production Area hatchery coho salmon smolts plotted by the year the smolts entered the ocean, expressed as percent smolt to adult survival (2a) and as the deviation from the mean survival transformed to the natural log of marine survival (2b).

Figure 3. Relationship between recruits/spawner and spawners for the 1990-1996 brood Tenmile Lakes coho salmon

Figure 4. Relationship between survival from egg to parr and seeding density of coho.
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In terms of developing biological criteria, probably the most important implication of these recruitment characteristics is that a population's ability to produce an increasing number of recruits per spawner as overall spawner numbers drop is more important to the continued existence of the population than spawner abundance itself. In other words, if the recruits per spawner increase only slightly or remains flat as the number of spawners decrease, the population may be at serious risk when marine conditions are unfavorable. In contrast, if the total number of spawners is low, but the population has retained the ability to produce large numbers of recruits per spawner when spawner density is low, then the likelihood that the population will persist and even rebound is high.

Revising the Low Abundance Paradigm for Coastal Coho
The marine conditions encountered by the offspring of coho that spawned from 1990 to 1996 were very unfavorable, as has been previously noted and illustrated in Figure 2. Associated with this unprecedented 7-year period of relatively stable but extremely low smolt to adult survival was an initial dramatic decline of coho abundance followed by relative stabilization at low abundance, in spite of continued low smolt to adult survival. Reductions in harvest rates during this helped minimize the adverse impact on survival (Figure 5). It had been previously thought that most coho populations, if faced with such poor ocean conditions and low abundance, would invariably decline. Further, it was thought that if such conditions existed for a sufficient period of time this decline would result in the extirpation of most populations. However, the actual performance of Oregon coast coho populations during this period suggests that this paradigm was wrong. Indeed, rather than continued decline, populations reached a low level of abundance and then stabilized at a new equilibrium.
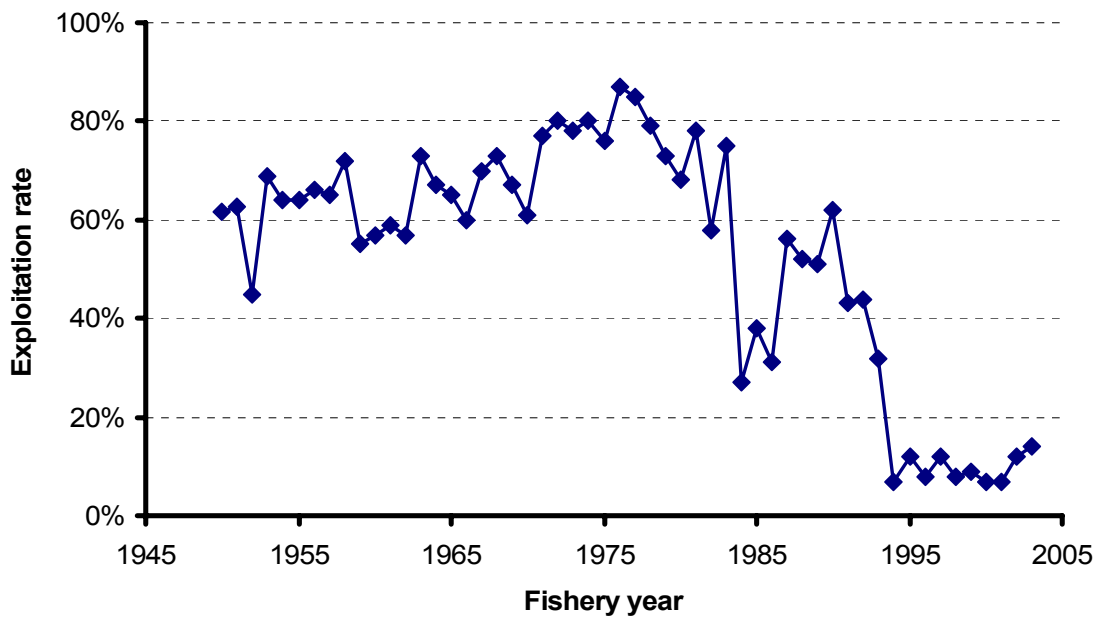

Figure 5. Estimated exploitation rates on Oregon coastal wild coho salmon in ocean fisheries,1950-2003.

The evidence for this condition can be demonstrated from an inspection of R/S values estimated for each population during this low survival period. For each population, R/S estimates for the 1990 to 1996 brood years were sorted in terms of parental (spawner) abundance from low to high. The R/S data were ranked within each population in terms of the corresponding spawner level, with "1" signifying the lowest spawner abundance, "2" the next lowest abundance, and so forth ending with "7" corresponding with the highest spawner abundance. This non-parametric treatment of the spawner abundance data provides a way to standardize spawner abundance among all populations to look at the relative relationship between spawner abundance and R/S on an ESU-wide basis. As illustrated in Figure 6, at the lowest relative spawner abundance (x-axis category '1'), R/S was greater than replacement (Ln(R/S) > 0) for all but one of the populations. In other words, the lowest spawner escapement observed for each population during this 7-year period of poor ocean survival resulted in a ratio of R/S greater than replacement in all cases except one. At the second lowest abundance ranking, only two populations were less than replacement. In contrast, at the highest abundance (rank = 7), all but one population was less than replacement.
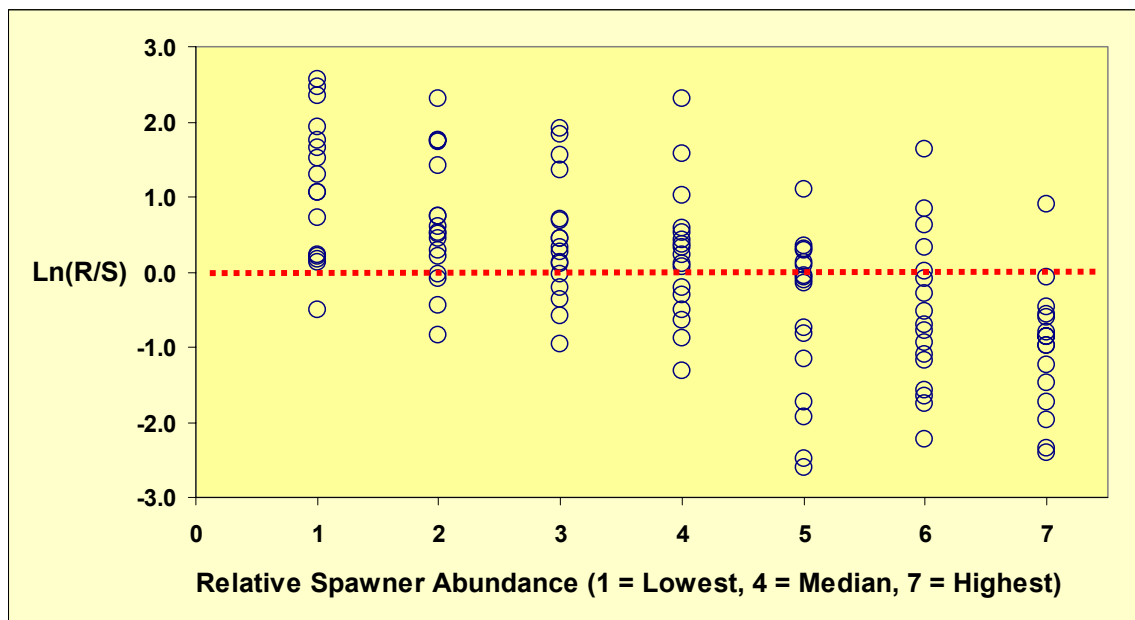
Figure 6 – Observed Ln(R/S) for 17 populations of Oregon coastal coho during a period of poor marine survival ordered by rank of ascending spawner abundance within each population; doted line represents spawner replacement line (R/S = 1).

These results are important for several reasons. First, even after experiencing the worst marine survival conditions of the 1990's, the populations still had the resilience to rebound from the lowest abundance levels. Second, the manner in which R/S values were symmetrically distributed with respect to relative spawner abundance suggests a system in balance, even though this balance occurs at extremely low spawner levels. Third, elevated productivity at low abundance indicates these populations had not

declined to levels where depensation was evident, as expected when populations near extinction levels. The old coho low abundance paradigm would have predicted that at such low marine survival rates nearly all of the R/S values would have been less than replacement and that as a consequence all populations would have been in decline. The data do not support this expectation.

Therefore, a new paradigm must be considered. It appears that the deterministic process for these coho populations under an extended period of poor marine survival as observed for the 1990-96 brood years is not one of continuing decline, but rather one of eventual stabilization around a new equilibrium. This result is consistent with the predictions made by Nickelson and Lawson (1998) in their earlier modeling of Oregon coast coho populations. This stabilization occurs because low spawner escapement resulting from poor marine survival is apparently offset by a strong density-dependent increase in freshwater survival of their progeny when spawner density is low. Therefore, the risk of extinction during these periods is related to the stochasticity of the recruitment process (i.e., populations at low abundance levels are more vulnerable to normal and catastrophic variations in recruitment than when abundance levels are higher). Clearly, coho populations are at greater risk at low abundance levels than when these levels are higher. However, it is the degree of random variations in the recruitment process, not an inevitable decline that is the reason for this increased risk.

Description and Documentation of Population Data

Data sets were developed for 17 populations of Oregon coastal coho. These data sets included spawner abundance estimates, distribution, and proportion of naturally spawning hatchery fish. Further, marine survival rates and estimated fishery mortality impacts were developed for the entire ESU because at present it was not possible to provide this information at the individual population level.

Spawner abundance of wild coho salmon in Oregon coastal streams has been estimated annually since 1990 using stratified random surveys (Jacobs and Nickelson 1998). Surveys were originally designed to estimate abundance at the ESU-wide level. Therefore, when the results from these surveys were used to estimate spawner abundance at the level of individual populations, the confidence levels associated with these estimates become much wider than those for the entire ESU considered as a single unit. With advent of the Oregon Plan for Salmon and Watershed in 1997, the sampling rate for survey sites was doubled from 240 to 480 sites surveyed to help remove some of the uncertainty associated with the population-level estimates. Further implementation of the Oregon Plan monitoring in 1998 resulted in the adoption of an integrated rotating panel sampling design developed by Stevens (1997) implemented as EPA's GIS-based Environmental Monitoring and Assessment Program (EMAP) site selection procedure (Jacobs et al. 2002.

Prior to 1998, the number of hatchery fish on the spawning grounds was estimated from the proportion of recovered carcasses found to have "hatchery" scale patterns. The scale analysis was not a perfect method for making hatchery/wild classifications for a variety of reasons, including the problem of obtaining an adequate number of reference samples

from known wild fish. It is possible for wild fish, particularly those from a lake system, to have scale patterns similar to hatchery fish. However, beginning with adults returning in 1998, almost all hatchery fish had been marked prior to their release with adipose fin clips. Therefore, the potential for misclassification error was reduced significantly. However, there were some cases (e.g., Nehalem) where estimating the percentage of hatchery for the entire population was confounded by the distribution of stray hatchery being concentrated in small portions of the basin. In these cases where the distribution of naturally spawning hatchery fish was non-uniform, an attempt was made to sub-divide the basin into smaller units, determine the proportion of hatchery and wild fish for each of these smaller units and then recombine these estimates in manner that was proportional to number of stream miles for each sub-unit represented. This recombined value was then used represent the percentage of stray hatchery fish for the entire population.

The impact of ocean fisheries during the period 1990 to 2003 was mortality caused by directed harvest, incidental mortality associated with other fisheries, and post-release mortality associated with the selective coho fisheries of recent years. Fishery related impact rates on wild coastal coho were based upon estimates provided by ODFW in the form of unpublished data summaries (Curt Melcher, ODFW, personal communication).

Spawner abundance estimates, percent of potential stream miles occupied by spawners, percent hatchery strays, and the number of subsequent adult (pre-harvest) recruits were summarized for the 1990 to 2003 brood years for 17 coastal coho populations. These data are provided in Table 2.

Table 2. Primary data used to evaluate the status of 17 populations of Oregon coast coho, summarized by brood year (1990 to 2003).

Brood year	Total spawners	Percent hatchery	Miles occupied	Wild recruits	Brood year	Total spawners	Percent hatchery	Miles occupied	Wild recruits
		Necanicum					**Nehalem**		
1990	183	0.0%	60	1,248	1990	4,843	86.2%	278	5,486
1991	1,089	0.0%	60	414	1991	9,571	30.9%	417	3,235
1992	169	0.0%	60	201	1992	3,359	38.6%	353	2,168
1993	720	0.0%	60	670	1993	11,494	72.5%	441	1,101
1994	386	0.0%	50	162	1994	4,743	36.4%	378	1,296
1995	176	0.0%	40	1,033	1995	3,258	41.7%	379	2,000
1996	614	0.0%	60	557	1996	3,778	73.3%	395	3,705
1997	142	0.0%	45	528	1997	2,352	51.7%	347	15,356
1998	952	0.0%	53	4,612	1998	2,171	15.1%	264	24,568
1999	515	0.0%	44	2,717	1999	3,821	10.4%	370	20,520
2000	489	0.0%	53	3,564	2000	14,346	0.8%	359	37,025
2001	4,574	6.6%	53		2001	24,880	8.6%	426	
2002	2,458	3.1%	60		2002	21,651	16.9%	456	
2003	3,051	0.0%	53		2003	32,716	3.1%	476	
		Tillamook					**Nestucca**		
1990	265	0.0%	90	1,490	1990	189	0.0%	109	695
1991	3,000	0.0%	210	700	1991	728	0.0%	131	336
1992	261	0.0%	157	330	1992	684	0.0%	119	2,067
1993	1,821	52.8%	151	721	1993	401	0.0%	75	566
1994	924	29.4%	214	443	1994	313	0.0%	87	310
1995	761	62.0%	154	293	1995	1,811	0.0%	112	256
1996	768	13.9%	161	2,157	1996	519	0.0%	109	2,551
1997	421	7.8%	119	1,871	1997	271	0.0%	62	1,271
1998	288	6.3%	67	1,529	1998	238	0.8%	26	4,438
1999	2,141	6.9%	138	15,659	1999	2,357	0.0%	90	18,731
2000	2,250	22.9%	135	16,404	2000	1,269	7.2%	92	11,961
2001	1,824	22.4%	207		2001	4,371	6.0%	137	
2002	14,022	2.1%	219		2002	16,753	1.9%	131	
2003	14,187	1.0%	221		2003	10,303	0.6%	155	
		Salmon					**Siletz**		
1990	434	11.3%	37	631	1990	441	0.0%	153	693
1991	39	0.0%	36	115	1991	3,365	70.8%	193	1,288
1992	143	80.4%	41	242	1992	3,340	26.7%	209	693
1993	364	0.0%	33	296	1993	400	0.0%	144	832
1994	1,554	93.1%	41	271	1994	1,200	0.0%	209	384
1995	1,325	84.0%	34	100	1995	607	0.0%	139	344
1996	2,703	90.0%	41	134	1996	763	0.0%	167	1,314
1997	417	43.2%	41	-	1997	336	0.0%	139	3,654
1998	432	78.7%	37	-	1998	357	11.2%	134	1,428
1999	173	28.3%	37	665	1999	1,364	11.0%	199	2,035
2000	394	100.0%	41	138	2000	3,387	0.0%	185	9,484
2001	877	100.0%	38		2001	2,454	46.1%	188	
2002	1,108	47.4%	37		2002	2,504	28.7%	209	
2003	1,738	93.2%	34		2003	8,421	3.6%	209	

Table 2. continued.

Brood year	Total Spawners	Percent hatchery	Miles occupied	Wild recruits	Brood year	Total Spawners	Percent hatchery	Miles occupied	Wild recruits
		Yaquina					**Beaver**		
1990	613	37.8%	114	951	1990	88	0.0%	15	457
1991	380	0.0%	89	2,627	1991	323	0.0%	11	362
1992	633	0.0%	120	6,470	1992	616	0.0%	22	352
1993	549	0.0%	111	5,591	1993	264	0.0%	7	559
1994	2,448	0.0%	133	439	1994	337	0.0%	11	459
1995	5,668	0.0%	133	621	1995	308	0.0%	11	350
1996	6,104	16.0%	122	2,774	1996	513	0.0%	11	2,661
1997	529	27.4%	115	687	1997	402	0.0%	22	662
1998	644	11.0%	114	3,553	1998	1,293	75.0%	22	3,180
1999	2,563	0.0%	113	27,138	1999	2,459	0.0%	17	5,971
2000	637	0.0%	100	19,257	2000	614	0.0%	22	5,088
2001	3,589	8.3%	128		2001	4,331	32.0%	22	
2002	23,800	0.0%	133		2002	5,470	4.3%	22	
2003	16,484	0.0%	133		2003	4,355	0.0%	22	
		Alsea					**Siuslaw**		
1990	1,197	0.7%	182	1,856	1990	2,685	0.0%	394	7,673
1991	1,561	0.0%	201	1,372	1991	3,740	0.0%	462	3,439
1992	8,487	17.2%	216	777	1992	3,440	0.0%	483	6,951
1993	1,071	0.0%	175	1,785	1993	4,605	3.8%	483	8,315
1994	1,279	0.0%	170	777	1994	5,164	37.9%	407	763
1995	681	0.0%	179	61	1995	6,089	0.0%	533	1,177
1996	1,637	0.0%	166	1,686	1996	10,333	26.2%	491	3,024
1997	928	26.7%	164	3,628	1997	668	0.0%	235	7,046
1998	1,732	96.8%	159	3,119	1998	1,327	18.2%	333	11,454
1999	2,071	24.8%	212	10,384	1999	3,193	12.5%	371	63,078
2000	3,363	0.0%	169	12,011	2000	6,532	0.0%	425	33,840
2001	3,920	26.3%	199		2001	10,606	0.0%	513	
2002	9,254	1.6%	227		2002	55,695	0.7%	515	
2003	10,281	0.0%	224		2003	29,059	0.3%	563	
		Lower Umpqua					**Upper Umpqua**		
1990	1,011	7.1%	223	10,486	1990	3,119	53.2%	595	3,814
1991	1,951	9.1%	241	2,602	1991	4,950	57.1%	661	3,664
1992	1,657	7.0%	255	10,381	1992	5,084	59.4%	315	6,853
1993	6,193	2.3%	326	6,761	1993	4,174	47.3%	480	5,461
1994	2,656	8.7%	286	1,189	1994	3,782	9.7%	570	1,663
1995	9,158	0.7%	360	4,572	1995	6,842	12.3%	636	4,107
1996	7,086	12.5%	273	3,004	1996	9,271	46.0%	906	5,128
1997	1,154	9.8%	242	5,135	1997	2,707	46.2%	591	7,728
1998	4,215	0.0%	285	12,945	1998	8,750	56.7%	562	24,170
1999	2,935	5.4%	316	24,100	1999	7,156	33.8%	677	20,299
2000	4,865	2.2%	295	17,121	2000	15,175	52.8%	524	17,153
2001	13,996	14.4%	336		2001	40,430	44.6%	785	
2002	21,717	2.7%	354		2002	24,738	28.0%	817	
2003	14,711	0.4%	338		2003	23,424	37.3%	764	

Table 2. continued.

Brood year	Total Spawners	Percent hatchery	Miles occupied	Wild recruits	Brood year	Total Spawners	Percent hatchery	Miles occupied	Wild recruits
		Siltcoos					**Tahkenitch**		
1990	1,622	0.0%	52	6,277	1990	1,085	0.0%	15	1,653
1991	2,895	0.0%	52	1,530	1991	1,215	0.0%	15	1,139
1992	391	0.0%	52	5,134	1992	318	0.0%	15	1,857
1993	3,622	0.0%	52	5,207	1993	954	0.0%	15	1,774
1994	1,426	0.0%	52	3,030	1994	1,062	0.0%	15	2,122
1995	4,497	0.0%	52	3,386	1995	1,627	0.0%	15	3,055
1996	4,775	0.0%	52	3,051	1996	1,627	0.0%	15	4,079
1997	2,653	0.0%	52	4,137	1997	1,858	0.0%	15	684
1998	3,122	0.0%	48	5,512	1998	2,817	0.0%	15	3,808
1999	2,819	0.0%	45	5,415	1999	3,769	0.0%	15	3,976
2000	3,835	0.0%	43	7,743	2000	634	0.0%	15	3,742
2001	5,104	0.0%	42		2001	3,526	0.0%	15	
2002	4,749	0.0%	43		2002	3,487	0.0%	15	
2003	6,628	0.0%	52		2003	3,203	0.0%	15	
		Tenmile Lakes					**Coos**		
1990	1,687	0.0%	38	9,650	1990	2,273	0.0%	131	26,485
1991	3,141	0.0%	38	3,599	1991	3,813	0.0%	172	15,756
1992	1,277	0.0%	38	5,813	1992	16,979	2.6%	203	11,816
1993	5,569	0.0%	38	7,734	1993	16,072	4.9%	194	13,226
1994	3,354	0.0%	38	4,674	1994	15,207	3.4%	196	1,287
1995	5,092	0.0%	38	5,606	1995	10,447	0.9%	192	3,238
1996	7,092	0.0%	38	6,627	1996	12,128	0.0%	167	5,214
1997	4,092	0.0%	38	8,930	1997	1,127	0.0%	149	5,074
1998	5,169	0.0%	38	11,921	1998	2,985	0.0%	159	35,587
1999	6,123	0.0%	38	15,805	1999	4,818	0.0%	203	37,706
2000	8,278	0.0%	38	7,313	2000	4,704	0.0%	175	29,964
2001	11,039	0.0%	38		2001	34,259	3.8%	211	
2002	13,861	0.0%	38		2002	33,265	0.6%	217	
2003	6,260	0.0%	38		2003	25,950	1.2%	203	
		Coquille							
1990	2,712	0.0%	225	12,796					
1991	5,651	0.0%	263	5,402					
1992	2,115	0.0%	199	2,416					
1993	7,551	2.2%	243	17,632					
1994	5,119	1.6%	238	6,533					
1995	2,116	0.0%	229	2,616					
1996	16,169	0.0%	253	2,886					
1997	5,720	0.0%	213	6,745					
1998	2,412	0.0%	195	13,967					
1999	2,667	0.0%	199	8,696					
2000	6,253	0.0%	233	26,221					
2001	15,665	17.4%	289						
2002	7,866	3.1%	259						
2003	22,565	0.5%	277						

V – Population Criteria Description and Rationale

Abundance Criteria

The development of biological criteria for coastal coho started with a consideration of the five attributes described earlier: abundance, productivity, distribution, diversity, and persistence. However, as various attempts were made to craft measurable and meaningful abundance criteria, it became clear that acceptable abundance levels can and will vary dramatically in a viable coho population based on variable marine survival and density-dependent freshwater survival. For example, low abundance during unfavorable ocean conditions may be a more "viable" abundance than only moderate abundance levels during favorable ocean conditions. Also, establishing minimum abundance levels necessary to avoid extinction was not informative about the conservation status of each population because of density dependent resilience in population productivity at low abundance.

For example, it was assumed that population abundance level immediately prior to a major downturn in marine survival, as occurred in the mid-1990s, was a critical factor in predicting whether or not a population would be lost (extirpated) during an extended period of poor marine survival. However, it became evident after testing this assumption with a variety of models and conducting a closer inspection of the actual abundance and productivity data, that starting abundance had very little effect on whether or not a population would survive a period of poor marine survival. Except when abundance levels are extremely low (in general much lower than what has been observed to date), the number of spawners in a population of coastal coho is a metric that in itself appears to have very little utility for the purposes of evaluating viability. Nickelson and Lawson (1998) presented results from their viability modeling of Oregon coastal coho that essentially agree with this perspective – spawner abundance by itself is generally a poor means to track the viability of coastal coho populations. Therefore, the importance of the abundance attribute – with respect to numerical extinction risk – was essentially excluded from consideration as a pivotal biological criterion for coastal coho.

Population viability criteria were thus developed to focus on four key elements: 1) productivity at low spawner density, 2) probability of long-term persistence, 3) within population distribution, and 4) within population diversity. The rationale and description of the test metric for each of these, as well as the method used to calculate the test metric, will be discussed individually in the following paragraphs.

Criteria 1 and 2 – Productivity at Low Spawner Density

As discussed previously, Oregon coastal coho populations appear to have considerable resistance to downward trends in abundance at low spawner densities. The productivity criterion was designed as a test to confirm that individual populations were performing in accordance with this expectation. For a coho population to display this resilience at lower abundance levels, it is necessary for the number of progeny (recruits) be equal to, or greater than, the number of parents (spawners) that produced them. In other words, when abundance is low, the ratio of recruits to spawners must be greater than one (i.e.,

replacement). Therefore, the basic data used to test this criterion are the observed ratios of recruits per spawner (R/S) during some test period when the spawner density was low. As discussed earlier and illustrated in Figure 3, R/S appears to vary widely and in a manner that is at least moderately dependent on spawner density. Therefore, to predict how a coho population might behave when abundance levels approach the extinction threshold, it was imperative that the forecast be inferred from population performance observed in the past when spawner densities were low. Population recruitment during periods of high spawner density is unlikely to be informative and, if relied upon for viability assessment, may yield predictions that overstate extinction risk.

The estimation of R/S is complicated by three factors: the presence of naturally spawning hatchery fish in several of the populations, the mortality impacts of the ocean fishery, and the effect of large annual variations in marine survival rates.

Regarding the first factor, hatchery produced fish are found in several populations of coastal coho, most notably the Nehalem, Salmon, and Umpqua. It is unknown how effective these hatchery fish are relative to wild fish in producing surviving progeny under natural conditions. However, a number of studies suggest that hatchery produced fish spawning in the wild are less likely to produce as many surviving offspring as wild spawners (Leider et al. 1990; Reisenbichler and Rubin 1999; Kostow et al. 2003; and McLean et al. 2003). Further, (Nickelson 2003) provided evidence that even if naturally spawning hatchery fish produced no surviving offspring, the mere presence of hatchery smolts within the basin may attract predators and reduce the productivity of the wild population

However, regardless of their reproductive success to the adult stage, Leider et al. 2003, and Kostow et al. 2003 found that if the relative proportion of hatchery fish in the natural spawning was high enough, their offspring would be the major component of the subsequent cohort of naturally produced smolts. Essentially, the reproductive inefficiency of the hatchery fish was countered if their proportion on the spawning grounds relative to wild fish was high. This situation makes the assessment of populations difficult because it is not clear if hatchery spawners are "propping up" the abundance of natural population. In addition, the fact that both the aforementioned authors found that naturally-produced smolts from hatchery parents survived poorer to the adult stage than offspring from wild parents suggests that conversion of natural spawners to returning adults is less efficient in basins where a large proportion of the spawners are hatchery fish.

Because of the uncertainty surrounding these issues and the range of different types of hatchery stocks being released, all fish that spawned naturally were counted equally as spawners and used in the denominator for the R/S calculation. The R/S numerator, recruits, was based upon counts of naturally produced fish only. Therefore, if hatchery fish, either through genetic or ecological interactions, were responsible for an adverse impact on the overall natural population, this effect should be evident in the estimated R/S ratio. This approach lets the performance of the fish provide the index of hatchery fish impacts, not a particular assumption concerning the relative reproductive success of naturally spawning hatchery fish and their ecological interactions.

The second consideration in defining a measurement for R/S was the impact of the ocean fishery. The issue is twofold. First, fishery impact rates on wild coho vary by year. Although in recent years this overall impact has been relatively low (less than 15% mortality), in the past it has been much higher (Figure 5). The full biological potential of a population in terms of recruitment from any year's spawner escapement is the estimate of returning adults prior to fishery impacts. Because the estimation of adults for each population is done on the spawning grounds, pre-fishery abundance is estimated by dividing each year's spawners estimate by the corresponding year's fishery survival rate (fishery survival rate = 1- estimated fishery mortality rate). This calculation was used to produce the value for recruits for each brood year estimate of R/S.

The second fishery issue related to the current harvest management framework used by the PFMC. Even under the most aggressive fishery harvest restrictions, reducing the impact on wild coho to less than about 8% from a management practically is extremely difficult because of incidental catches in other fisheries (primarily Chinook) (Sharr et al. 2000). Therefore, calculating an R/S measurement of 1.0 would not mean replacement was being achieved, because a portion of the return would be removed by the fishery (recall that "R" in R/S is a pre-fishery estimate of adult abundance). A measured R/S value of 1.0 would translate into effective R/S on the spawning grounds of 0.92. As a precautionary accommodation of this situation the standard for R/S to be met under poor marine survival conditions and low parental spawner density was set at 1.1. Therefore, a value for R/S of 1.1 would ensure that, in terms of the recruits that actually reached the spawning grounds, the population had reproductively replaced itself.

The third consideration in defining a measurement for R/S was the variability in marine survival conditions. Estimates of R/S for years when marine survival was very poor have the potential to yield values 1/10 of those that might be estimated when the marine survival was very high. It is difficult to know how data should be interpreted if marine survival was variable. Two approaches were taken to address this problem. In the first, only those R/S data obtained during a period of similar marine survival conditions were used for the biological criteria. In the case of this evaluation, this restricted the analysis to a period of exceptionally low marine survival rates impacting the offspring of fish that spawned in 1990 to 1996 (brood years). The second approach to the problem was to standardize the number of adult recruits to the marine survival rates observed during this low period (1990-96). This was done for each brood year after 1996 by multiplying a marine survival rate adjustment factor by the number of recruits observed. Essentially, this adjustment standardizes the recruit returns to what they would have been had they experienced the same marine survival conditions as did fish belonging to the 1990-96 broods. This R/S adjustment can be described in terms of the following equation:

$$R^*/S = [(R)(OPImean/OPIyr)] / S \qquad (1)$$

Where R^*/S = R/S estimate for a particular brood year adjusted by observed marine survival rate, OPImean = average Oregon Production Index (OPI) marine survival rate

for the 1990 to 1996 brood years, and OPIyr = OPI marine survival rate for the brood year to be adjusted.

Application of Criterion 1 as one test of a population's biological health is essentially to provide the answer to the question: what is the probability that the mean R/S during years of low spawner density was statistically less than an R/S value of 1.1? If this probability is sufficiently high, then the population is at risk of not replacing itself during periods of poor marine survival conditions and is therefore at risk of extinction. Alternatively, if the probability is low then the population appears to have the resiliency necessary to avoid extinction.

To provide an estimate of this probability the following protocol was developed. First, all R/S data for brood years after 1996 (1997 to 2000) were adjusted for marine survival using Equation 1. The data were then divided into an unadjusted data set based upon the results for the 1990 to 1996 brood years (n = 7), and an expanded data set based upon the 1990 to 2000 (n = 11) brood years which included the marine survival adjustments to R/S. Both data sets were sorted by spawner density and only those R/S data corresponding with low spawner densities were selected for comparison. In the case of the unadjusted data set, the selection threshold was 5 spawners per stream mile. For the expanded data set the selection threshold was 15 spawners per stream mile. Test data were transformed from R/S to natural log(R/S) and the mean of these values evaluated using a 1-tailed, t-test to determine the probability that the data points were drawn from a distribution of Ln(R/S) values that had a true mean ≤ the natural log of 1.1 (replacement).

The test of the unadjusted data set was defined as "Productivity Criteria 1" or "Pro1". The test of the expanded data set was defined as "Productivity Criteria 2" or "Pro2". As a metric Pro1 was a more direct measurement of low density productivity. Pro2, because of the expanded range of data points (sorting threshold of 15 spawners per mile) and adjustments for marine survival for R/S after 1996, was expected to be more variable and sensitive to error than Pro1.

A range of values were considered as low density thresholds for the data sorting and selection step of the productivity protocol. The problem was that a threshold set too high runs the risk of picking spawner densities that do not reflect how a population will truly respond when the critically low spawner levels are observed – generally thought to be less than four spawners per mile (Sharr et al. 2000). In other words, the R/S values are not representative of the critical spawner density range. The value of 15 fish per mile was based upon an average estimate of the density of spawners needed to fully seed the population's habitat. For some populations, such as the upper Umpqua, subsequent recruitment modeling suggested that this threshold was indeed too high. However, there was a desire to not use population specific estimates of full seeding to set the threshold for the productivity because to do so would require the reliance on the same spawner-recruit model as used for the persistence and diversity criteria. This would have been inconsistent with attempting to make the methodology associated with each criterion as independent as possible. However, the reader should be aware that results for the Pro2 criterion will be underestimates of the population's low abundance recruitment potential.

In contrast sorting threshold that are too high, setting the sorting threshold too low runs the risk of having no data to work with because all of the spawner densities end up being greater than the threshold. As subsequent analysis demonstrated, the majority of populations fell into this category in that all of the observed spawner densities were greater than 5 fish per mile. Although this does render the computation of the test metric impossible, it does suggest that the populations were productive enough during this period of low marine survival that they never declined to a 5 fish per mile density. This result in itself is supportive of the population being viable.

 The coastal lake populations, however, presented a special problem in setting a sorting threshold for Pro2. There was only one R/S point in data for all 3 lake populations that was less than 15 spawners per mile. The lake populations, apparently because they have a much higher ratio of over-winter habitat (the lakes) to spawning habitat, typically have spawner densities 3 to 5 times greater than all other populations in the ESU. Therefore, the definition of "low spawner density" for the lake populations, and thereby the sorting threshold for the Pro2 criteria was set at 75 fish per mile with the intent that this would possibly bring more data into the analysis and still be consistent with the biological characteristics of these populations. It is stressed that this higher sorting threshold should not be interpreted that these lake populations are inherently more at risk of extinction.

Criterion 3 – Long-term Persistence

A wide variety of viability models have been used by conservation biologists to estimate the vulnerability of populations to extinction (Shaffer 1981, 1990; Murphy et al. 1990; Nickelson and Lawson 1998). This analysis combines a deterministic recruitment model and a simulation of random fluctuations in environmental conditions to forecast future population abundance. However, more than one forecast is used per population, because the inclusion of random variation in the recruitment process ensures that each forecast will differ slightly from another. It is not unusual in this type of analysis to perform 5,000 forecasts for each population. To summarize such results, the proportion of all of the forecasts that yield a prediction of extinction is calculated. This extinction proportion is then used as an estimate of the probability of extinction for the population. Obviously, populations with a low probability of extinction risk are healthy and are likely to persist.

Recruitment Model - Nested within the viability model used to forecast persistence for coastal coho is a recruitment model that is the basic 'engine' by which spawners in any given year are converted into adult recruits that return 3 years later. The recruitment dynamics of coastal coho was assumed to best correspond with the Ricker recruitment function as illustrated in Figure 7 and described by the equation:

$$R = S \, e^{\,(a + BS)} \tag{2}$$

where R = the total number of adults (pre-harvest) produced from the spawners 3 years previously, S = number of spawners, e = base of natural logarithms, a = parameter for intrinsic population growth rate, and B = parameter inversely proportional to habitat capacity.

While other recruitment functions were considered for use with coastal coho, for example Beverton-Holt (Beverton and Holt 1957), and hockey stick (Barrowman and Myers 2000 the Ricker model was selected for use because this model seemed to best fit the actual recruitment data for Oregon coastal coho. As information presented earlier suggest, the R/S values obtained for Oregon coast coho populations show an inverse relationship with spawner abundance (i.e., as spawner abundance trends down, R/S trends upward). This pattern is inconsistent the fixed R/S assumption of the hockey stick model for spawner abundance less than full seeding. The Beverton-Holt recruitment curve was not selected because a preliminary scoping of the data indicated that better fits were obtained with the Ricker function. Zhou (2000) also came to this conclusion in his selection of the Ricker model over the Beverton-Holt model for his analyses of coastal coho in Siltcoos, Tahkenitch, and Tenmile lakes.

Some have suggested that when population recruitment data containing substantial measurement error are fit to the Ricker model the resulting recruitment curve will be biased and overestimate R/S at low spawner density (Caputi 1988; Quinn and Deriso 1999). However, Kehler et al. (2002) found that, in general, low density R/S was not over-estimated in the fitting of 'noisy' data to the Ricker recruitment model. . The problem however, for the Ricker model is that once the level of spawners needed to achieve full seeding of the habitat is reached, additional increases in the number of spawners results in a proportional decline in the number of recruits produced (Figure 7). Under extremely high escapement levels the Ricker model predicts only a few recruits will be produced. This is biologically unreasonable and not supported by what little recruitment data there are from exceptionally large spawner numbers. Therefore, for the purposes of modeling viability the form of the Ricker curve used in computer simulations assumed a leveling out of recruits for spawner abundance greater than Smax as illustrated in Figure 7.

As discussed in Section IV, variation in marine survival has a strong influence on the number of coho returning to spawn in any particular year. As a consequence the relationship between spawners and recruits will be heavily modified by variations in marine survival. Therefore, marine survival was specifically added to the recruitment function as a second variable. Incorporating this new variable into Equation 2 yielded the following:

$$R = S\,e^{(a + BS + cO)} \tag{3}$$

where, c = additional parameter for marine survival and O = a standardized measure of marine survival calculated from OPI (Oregon Production Index) marine survival data as:

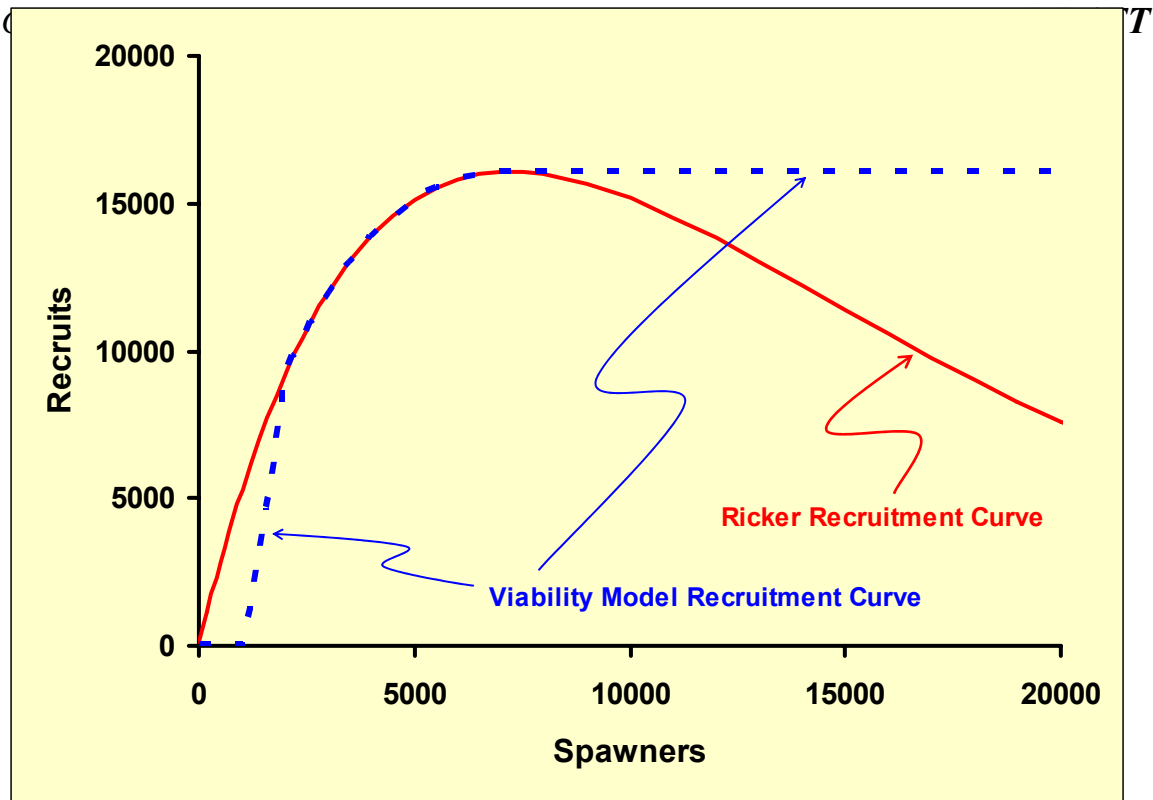$$O = Ln(OPI_{yr}) - Ln(OPI_{1958\text{-}2000mean}) \tag{4}$$

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Figure 7 – Generic example of Ricker recruitment curve and the modified version of this curve used to simulate recruitment in the viability model (see text).

It should be noted that although the spawner and recruit data for individual populations spanned a 14-year period from 1990 to 2003, marine survival, O, was standardized to the average OPI survival for the 1958 to 2000 broods. This was done primarily to facilitate viability model recruitment forecasts for the full range of marine survival rates observed during the entire period of record (1958 to 2000 broods).

For the purposes of estimating the recruitment equation parameters a, B, and c for each population, Equation 3 was rearranged to the following form:

$$Ln(R/S) = a + BS + cO \tag{5}$$

where 'a' becomes the intercept and 'B' and 'c' are the slopes for the variables, spawner escapement and ocean survival, respectively. Using equation 5, data were analyzed by multiple linear regression to parameterize values that best explained the observed variation in estimates of Ln(R/S). Because of the 3-year life cycle of this species, the available data series from 1990 to 2003 yielded only 11 brood years of data from which to estimate parameter values. This is a marginal number of points from which to obtain accurate parameter estimates. For example, Burgman et al. (1993) suggests that inaccurate parameters estimates are likely when the number of observations is less than 15.

To shed additional light on the possible implications of this problem, a separate recruitment analysis was made based on spawner density data collected from 10 standard survey sites of Oregon coastal coho. These data, which were collected from the 1950s to present (n = 46), consist of peak spawners per mile estimates of adult coho in specific, short sections of stream that were surveyed each year. For data from each of these 10 standard survey sites, Ricker recruitment parameters were estimated first using all of the data points (1958 to 2000 brood years) and then secondly, using data for only the 1990 to 2000 brood years. The reader will note that this second time series corresponds with the period of data collection for the full population abundance estimates and the associated concerns of this being an insufficient number of years (n = 11) to estimate accurate recruitment parameters.

For each survey stream, a comparison of parameter estimates from the short time series (1990 to 2000 brood years) versus parameter estimates from the longer time series (Figures 8, 9, and 10) did not yield significantly different values (paired t-test, P = 0.05) for the recruitment equation parameters 'a' (intrinsic growth rate) and 'c'(marine survival rate). However, for the habitat capacity parameter, Smax, the values estimated from the longer time series were statistically larger. It also appeared, that particularly with the marine survival parameter, c, the range of difference among populations was greater for the short time series as opposed to the longer time series.

In addition, the multiple regression results confirmed that the statistical confidence in parameters estimated from the longer time series was greater than from the shorter time series. For example, across the 10 locations examined, all but one of the regression coefficients was statistically significant for the long-term data set, whereas only about half of the regression coefficients for the shorter time series were significant.

From these results it may be reasonable to make the following inferences with respect to the recruitment parameters estimated for the populations evaluated in this study. First, as additional years of population abundance estimates accumulate and are included in the data set, the level of differences among populations, especially for the marine survival parameter, will likely decrease and the proportion of coefficients that are found to be statistically significant will increase. Second, although the parameter estimates for individual populations may be inaccurate, across all populations there is no evidence that using a shorter time series has caused an overall pattern of either strong positive or negative bias in the values for the 'a' and 'c' parameters, whereas there was some negative bias with respect to the 'Smax' parameter (i.e., the short-term data set yields smaller values than the longer term data set).
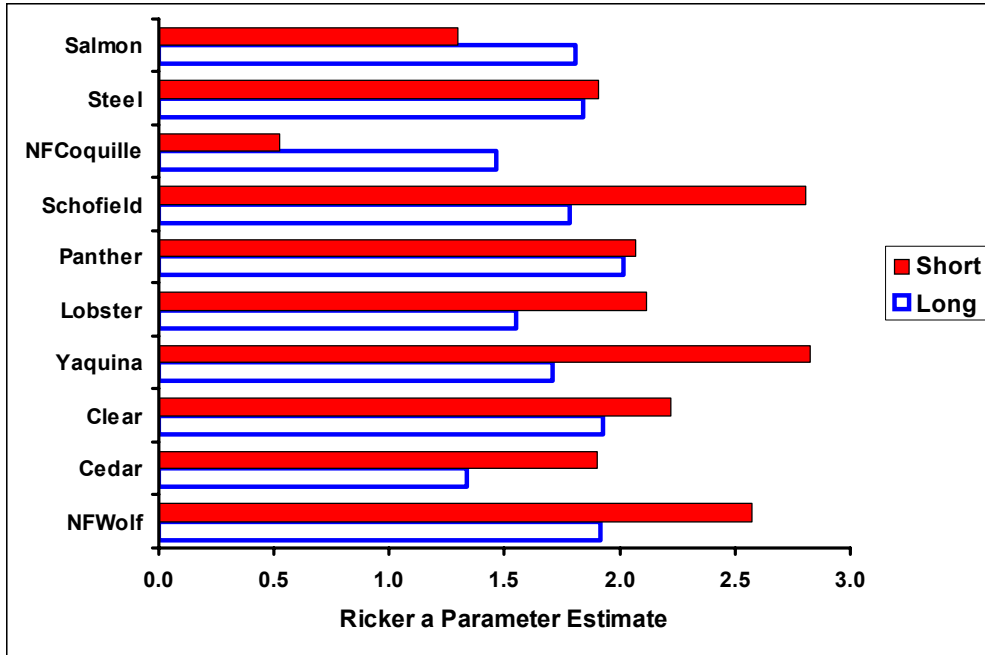
Figure 8. Ricker a parameter calculated from peak fish per mile counts on standard index streams for the 1958-2000 brood years (Long) and the 1990-2000 brood years (Short).
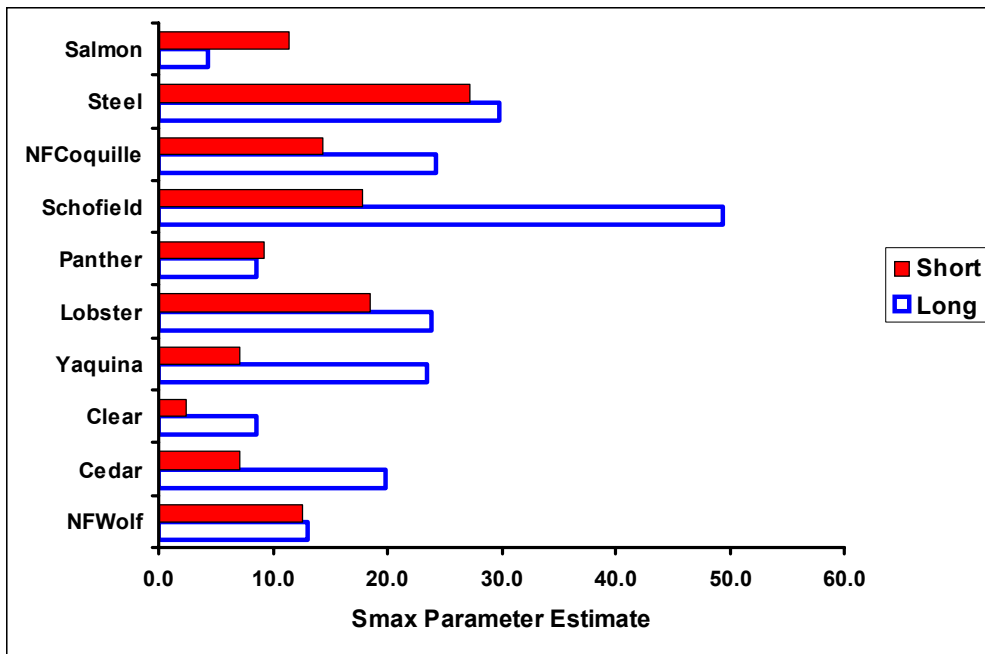

Figure 9. Smax calculated from peak fish per mile counts on standard index streams for the 1958-2000 brood years (Long) and the 1990-2000 brood years (Short).
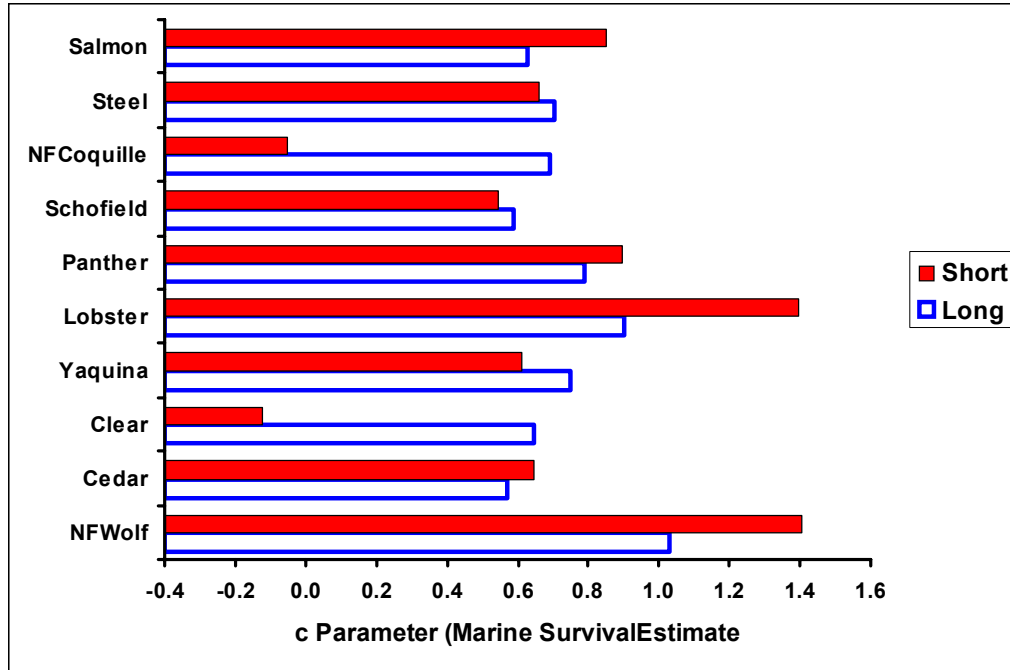
Figure 10. Marine survival parameter c calculated for standard index streams for the 1958-2000 brood years (Long) and the 1990-2000 brood years (Short).

Stochastic Simulation of Population Abundance for Test Period – To complete the development of the viability model, the recruitment 'engine' (Equation 3) and associated parameters estimated for each population were integrated into an Excel Basic macro that upon execution forecasts the likelihood of population extinction. An important part of this program, and its simulation of recruitment over a multi-generation time horizon, was the inclusion of expected environmental variation. To accomplish this Equation 3 was rewritten slightly to the form:

$$R = S \; e^{\,(a_{rand} + BS + cO_{yr})} \tag{6}$$

In this recruitment equation, 'a_{rand}' is substituted for 'a' and signifies that the parameter becomes a randomly picked number from a normal distribution having a mean equal to the estimated value for 'a' and a standard deviation equal to the standard deviation of the multiple regression estimated from the fitting of observed population data to Equation 5. For example, suppose the result of performing a multiple regression of spawner, recruit, and marine survival data for the 1990 to 2000 brood years of coho returning to a basin yielded a regression standard deviation of 0.70 and a value for the 'a' parameter of 2.2. Then the recruitment simulation would proceed such that each time recruits are forecast, the value for 'a_{rand}' in Equation 6 would be obtained as a random number selected from a normal distribution having a mean of 2.2 and a standard deviation of 0.70.

The minor notation change from 'O' to 'O_{yr}' for marine survival in Equation 6, signifies that for each year of the model run a different value for marine survival was assumed.

Since most of the model runs were made for a time period of 100 years (about 33 generations), it was necessary to select a sample of marine survival rates in order to model recruitment. This could have been done in manner similar to the process used to randomly pick values for the Ricker 'a' parameter (i.e., randomly draw a number from a normal). However, this approach has problems in that marine survival does not occur in a random pattern (Figure 2). For example, if poor marine survival is observed in one year, the likelihood is that a poor marine survival will also occur in the next year. For a coho population the challenge of surviving poor marine survival conditions is not one of dealing with the occasional rare event, but rather facing these rare events in a block of multiple years together. Clearly the latter situation would pose a much greater challenge to the survival of the population and it was this likely situation that the viability model was modified to simulate.

It is possible to correct for this autocorrelation problem within modifications to the structure of a viability model, using a variety of approaches including those proposed by Morris and Doak (2002). The approach used for coastal coho, however, was basically to make this modification using the observed marine survival rates, in the order that they occurred between 1960 and 2002, as a template for the future 100-year period modeled. Because the observation period covers only 43 years, it had to be replicated to provide enough points for the 100-year simulation of the model. Mechanically how the model prepared the sequence of marine survival rates for the recruitment calculations was to first artificially string together 4 replications of the marine survival rates observed from 1960 to 2002. Essentially replicating the pattern of Figure 2 four times to create a cyclic string of 172 values having 4 high and 4 low periods of marine survival. For each 100-year model test period a different starting place was randomly selected (from a uniform distribution) from the first 43 years of the 172-year string. The first 100 years after this starting point were used as the marine survivals for the model run.

When simulations are performed using the protocol described thus far, a problem occurs in that regardless of how low a spawner forecast is, it will always yield recruits in the next generation – even if the spawner forecast is a nonsensical value like 1 (i.e½., one fish). This makes it impossible for the model to yield an extinction result. Therefore, additional modifications were needed to discount recruitment when spawner numbers fell below some critical level. That coastal coho rarely experience extremely low, near zero, spawner levels there is no observational basis to predict or model their performance when they enter into this range. .

Although the Ricker recruitment model predicts maximum R/S at these low abundance levels, there are no data to confirm that this indeed occurs which is troublesome problem. . Especially so since there is evidence that at such low levels the expected recruitment mechanisms may fail (Glipin and Soule 1986). Although this phenomenon of low density failure has been much discussed, the empirical demonstration of its existence is rare. Regardless, it is hypothesized that either because of genetic problems or the inability of spawners to find mates when their density within the basin is low, the productive capacity of a population may irreversibly decrease as the population declines below some critical level of spawners. Using the terminology of Ginzburg et al (1982)

this irreversible threshold is defined as the quasi-extinction level or commonly abbreviated as 'QET'. In developing the viability model, a conservative assumption was made that this QET point existed for each population, even though direct observational evidence for this threshold was lacking.

To adjust the viability model to incorporate QET required that QET be determined for each population; a difficult task considering there were no data from which to base this determination. Soule (1980) suggested that based upon the adverse impacts of inbreeding depression at low abundance levels that a threshold of 50 individuals may be appropriate. A threshold of 50 has also been utilized for modeling purposes by the Williamate/lower Columbia and Interior Technical Recovery Teams (TRT). However, populations of coastal coho differ widely in the size of their home range. For example, the Necanicum population has 52 stream miles of habitat, while the Upper Umpqua population has in excess of 1000 stream miles of habitat. When these differences were converted into fish per mile, it seemed unlikely that the probability of finding a mate in the Necanicum (1.0 fish per mile) was at all similar to finding a mate in the Upper Umpqua (0.05 fish per mile). Therefore, as a precautionary approach it seemed reasonable that the QET for populations in large basins should be larger than for populations in small basins.

The QET level for each population was set at the equivalent of 1 spawner per average miles of stream occupied by spawners during the recent down turn in marine survival or 50 fish which ever is larger. The rationale for selecting these numbers was that the numbers were large enough to avoid the potential serious effect of inbreeding depression (i.e. $n > 50$); and they resulted in a uniform spawner density as the threshold regardless of the size of the basin (except for the very small basins that defaulted to a QET of 50). The QET values for each coastal coho population are presented in Table 3. In runs performed by the viability model, when spawner abundance was less than QET, the spawner abundance was reset to equal zero (effectively extinction) as a precautionary approach. Rather than have a knife edge transition from a high R/S to an R/S of zero at the QET threshold, the viability model was set to begin discounting R/S from the Ricker model expectations at a spawner level of twice QET. Below twice QET, incrementally smaller abundance levels were assigned incrementally smaller R/S recruitment values, in a linear fashion, such that when abundance reached QET, R/S had declined to zero.

As noted earlier, one possible shortcoming of Ricker recruitment function when applied to coho is that it will yield a decreasing number of recruits as the number of spawners increase past the spawner level needed to achieve maximum seeding. As a matter of terminology, this full seeding level is commonly abbreviated, Smax, and can be estimated as (-1/Ricker B parameter). In some cases, the rate of decrease in recruits at spawner levels greater than Smax is unrealistically steep. Mathematically, in these instances it is possible for a very high number of spawners to produce so few recruits that near QET levels will result. Since the model simulations are capable of generating these kinds of large spawner escapements, there was a desire to control the output such that it wouldn't produce the illogical result of extinction because of too many spawners. To accomplish this, a step was inserted into the program code to check spawner abundance before each use of the recruitment equation. If this check revealed a spawner abundance value

greater than Smax, then the spawner abundance was reset to Smax. The effective result of this change was to level out the recruitment curve at spawner densities greater than Smax – giving the overall curve perhaps the appearance of an inverted ski (Figure 7).

The program code for the persistence model also had a provision for assigning additional, density independent mortality to the predicted number of recruits. This capability was used to incorporate a fixed fishery impact rate of 8% for all model runs. For the similar rationale described earlier in the presentation of the Productivity Criterion, this fishing mortality addition was an attempt to reflect the most likely management condition during periods of poor marine survival. However, the ability to perform model runs with any post-recruitment mortality permitted the testing of the outcome to different assumptions about future marine survival conditions, future of the freshwater habitat conditions, and different fishing impact scenarios. Several such 'what if' model runs were analyzed for discussion in Section VIII.

Table 3 – QET values for populations

Population	QET	Population	QET
Necanicum	50	Lower Umpqua	280
Nehalem	370	Upper Umpqua	590
Tillamook	170	Coos	180
Nestucca	100	Coquille	240
Salmon	50	Siltcoos	50
Siletz	170	Tahkenitch	50
Yaquina	120	Tenmile	50
Beaver	50		
Alsea	190		
Siuslaw	460		

The Mechanics of the Model Runs – The model was run for each population to estimate the probability of extinction risk over a future 100-year time period. For each 100-year trial, the sequence of spawner escapements were inspected to determine if the average of any 3-year consecutive string of spawner abundance was less than the population's QET. If a 3-year average less than QET was found, then the trial outcome was classified as an extinction event. This is a conservative interpretation of extinction, as it takes only one occurrence of an average less than QET within a 100-year sequence to trigger the extinction flag. However, this is a standard that has been adopted by the OTRT and is also presented by NOAA Fisheries in their VSP document (McElhany et al. 2000).

For each population, a model run consisted of 3,000 of these 100-year trials. After all of the trials were completed, the probability of extinction was calculated the number trials that were classified as extinction events divided by the total number of trials (i.e., 3,000).

Criterion 4 – Within Population Distribution

Within the freshwater portion of a population's home range the manner in which juveniles and adults (spawners) are distributed is an important consideration in assessing the conservation status of a population (McElhany et al. 2000, Bisson et al. 1997). For example, a chance catastrophic event in a tributary watershed of a larger basin would have less impact on a population whose members are uniformly distributed the basin compared to population that is not widely distributed and has the misfortune of being concentrated in the tributary where the that catastrophic event takes place. Populations that are well distributed across potential freshwater distribution of spawner habitats can also be expected to demonstrate greater resilience to both spatial and temporal variability in habitat conditions (Williams and Reeves 2003). Both natural and anthropomorphic disturbances create a shifting mosaic of abiotic and biotic conditions across the range of salmon species that is reflected in patterns of distribution and life history attributes (Reeves et al. 1995). The approach used to developing criteria for diversity focuses on evaluating the spatial distribution of spawners within a population and their potential to maintain spatial structure during periods of poor ocean conditions.

In addition to the advantage of spreading risk, broad distribution of individuals throughout a basin also has benefits in terms of maintaining genetic adaptation and diversity. The genetic characteristics of a population are partially controlled by selective processes working to cause the adaptation to the local habitat conditions. Since the habitat conditions throughout a basin are not uniform it would be expected that a diversity of selection pressures and adaptation strategies potentially exist for coho throughout a basin. Therefore, a population that is widely distributed is most likely interacting with the greatest diversity of habitats and therefore may develop and maintain the greatest genetic diversity. This mechanism is probably more important for populations in large basins than for those in small basins because the likelihood of within population clustering (sub-populations) and differences in habitat are greater for large basins.

The utility of using distribution as a diversity metric also has a practical advantage in that the genetic characteristics that are important to adaptation and survival are extremely difficult to directly assess. Essentially, genetic traits that are adaptive and important to survival are largely cryptic. Therefore, distribution becomes a proxy for these invisible traits under that assumption that if full interaction between the heterogeneous habitat of a basin and population members is ensured, then selection pressures will enable the process of genetic adaptation and evolution to proceed, regardless of whether we are able to directly observe or measure it.

Distribution has another feature for coho salmon that makes it a valuable tool for status assessment. As described by Nickelson and Lawson (1998) and observed by our monitoring program, the distribution of spawners is dynamic from year to year depending on annual changes in life cycle survival, in particular marine survival conditions. When marine survival conditions are poor, the distribution of spawners collapses to the better habitat within the basin. However, when marine survival rates are high, coho adults

demonstrate the capacity to spread out and fill nearly 100% of the available habitat (Jacobs et al. 2002). Therefore, the dynamic contraction and expansion of coho distribution within a habitat is another good indicator of population function and status.

In developing a means to quantify coho distribution for use as a criterion, it was necessary to recognize that even healthy populations will experience periods when the distribution of spawners collapses and other times when this distribution expands. The challenge was to select a criterion that would identify when a collapse in distribution becomes greater than should be expected for a healthy population under poor marine survival conditions. Although the concepts important for describing diversity and distribution are well documented, neither specific nor general guidance for developing measurable criteria were found. The problem of identifying such criteria was described by Kareiva (1990) when he concluded that "simply saying that the spatial environment is important is to mouth a platitude". For coastal coho populations, distribution data obtained during the recent period of poor marine survival provided the opportunity to develop a metric that helps quantify the patterns of distribution and than can be established so as to be protective of overall population diversity.

Development of a metric with measurable criteria was possible because of the availability of comprehensive and spatially explicit spawner density information. These data represent approximately 250 survey reaches/ESU from 1989 to 1996 and approximately 475 survey reaches/ESU from 1997 to 2003. Multiple observations of coho spawners are made for each survey reach each year, allowing determination of both peak counts of spawner abundance and estimates of total abundance for the season expressed as spawners/mile.

Although primarily designed to provide spawner abundance estimates at the ESU and Monitoring Strata levels, these data also support analysis of spawner abundance levels within each sub-watershed that comprises the habitat of each coho population. Talabere and Jones (2002) used this approach to determine the location of sub-watersheds that supported spawner abundances greater than 4fish/mile during periods of overall low abundance.

We developed viability criteria for coho distribution based on these observations combined with the following assumptions:

1. Coho Populations are distributed across most of the available spawning habitat within the sub-watersheds that comprise the area occupied by the population.
2. Results from the ODFW spawner sampling protocols are good indicators of population distribution and are directly related to population diversity, spatial structure, and connectivity.
3. The diversity criteria, combined with analysis of the spatial pattern of distribution can be used to make inferences about coho life history diversity and inform discussions of genetic diversity.
4. The spatial pattern of areas with higher than average spawner densities <u>within</u> a population are appropriate for developing distribution criteria.

The distribution threshold was set at 50% occupancy of the spawning habitat. In other words, if less than 50% of the spawning habitat was occupied by spawning coho then the population would be classified as having an abnormally severe distribution collapse. Such a population would be considered at risk for the distribution criterion.

However, the interpretation of a 50% threshold is highly dependent on how the term "occupied" is defined. Given the sampling protocol for stream surveys, how frequent and concentrated do observations of spawners need to be before a stream or watershed should be classified as occupied? Further, is this definition for occupied workable for both large and small basins?

The first part of a solution to this challenge was addressed by partitioning each population's habitat into areas defined by the USGS 5th field Hydrologic Unit Code (5th field HU). The USGS HU system provides standardized, hierarchical classification of drainage networks. Most of the medium or larger coho populations exist in basins that are classified as 4th field HUs, whole watersheds such as the Nehalem or Alsea. These watersheds are in turn comprised of multiple 5th field and even smaller 6th field HUs. For the retrospective analysis, we used Talabere and Jones (2001) method to combine spawner data for each the 6th field HU over a twelve year period. The number of 6th field HUs within each population became the units by which the 50% determination was made. Only 6th field HUs with spawning habitat were considered for this evaluation. For example, if a population existed in a basin containing 12 HUs, then at least 6 of those had to be occupied in order to 'pass' the population distribution threshold. For continued analysis of distribution, we recommend using 5th field HUs as the unit of evaluation. This is because at the 5th field scale, as surveyed by ODFW's EMAP protocols (Stevens 2002), the sampling rate is sufficient to support an analysis of distribution at much shorter intervals than was done in the retrospective analysis (2-3 years compared to 6-11 years).

Having defined 5th field HUs as the unit of measurement, the problem of defining 'occupied' was now reduced to the level of these units. In other words, the metric needed to address the question, "what qualifies as an occupied HU?" It was decided that a feasible approach would be use the ODFW's existing EMAP survey sampling design as a means to systematically inspect a HUC for spawners. The EMAP sampling design is a protocol for selecting survey sites in a manner that is spatially balanced and drawn from within the known coho distribution. Based upon the data collected from these selected surveys sites, a 5th field HU was classified as occupied if for at least 50% of the sampling sites examined, the density of naturally produced coho was 4 fish per mile or greater. A minimum density of 4 fish per mile was selected on the basis of the spawner frequency distributions developed by Talabere and Jones (2001), and by work conducted by Sharr et al. (2000) that suggested that at densities less than this level, the probability of each spawner finding a mate began to decline. The 4 fish per mile threshold also is concordant with Pacific Fishery Management Council rule and policies, although that was not the basis for selection in this analysis.

Naturally spawning hatchery fish are not counted in the definition of whether or not a sampling site was occupied. Excluding hatchery fish was done to keep the interpretation of fish distribution data consistent with the intent of the distribution criterion. Specifically, to ensure that distribution measurements were not confounded by breaking the closed loop between the natural habitat and the fish that had exclusively interacted with this natural habitat.

Criterion 5 – Within Population Diversity

Within population diversity is the result of phenotypic differences among individuals. These differences provide the flexibility of the population as a whole to respond successfully to short-term environmental variations. They also are the basis by which populations are able to adapt and evolve as conditions within their home range go through changes that are more permanent. Therefore, maintaining sufficient within population diversity is an issue of both short-term and long-term survival. The development of this criterion was based upon the importance of diversity as a factor in evaluating the conservation status of a population.

Within population diversity is controlled by a variety of forces including: evolutionary legacy, immigration from other populations, mutation, selection, and random loss of genetic variation due to small population size. However, it is this last factor that is most commonly recognized as a concern for species that are vulnerable to extinction. The genetic consequences of small population size and numerous approaches to defining minimum population abundance thresholds have been investigated widely (Soule 1980; Lande 1995; Franklin and Frankham 1998; Rieman and Allendorf 2001). In nearly all cases, this becomes an exercise of identifying a rate at which genetic variation can be lost without a causing a risk to a population's short or long-term persistence. The diversity criterion was developed around this concept.

After considerable review of the genetics and conservation biology literature, it was concluded that maintaining at least 95% (or losing no more than 5%) of a population's heterozygosity over a 100-year period as recommended by Allendorf and Ryman (2002) would be used as a conservation threshold. In this formulation, heterozygosity is used as the metric to represent population diversity. However, having defined the threshold in genetic terms it was then necessary to have a method to actually measure or forecast this loss rate for coastal coho populations. For this purpose we used a long known, relationship between a theoretical population size and the rate at which heterozygosity is lost. This relationship, developed in the early in the 20th century by Wright (1969), can be described by the following equation:

$$\Delta H = (-1/2Ne)(100\%) \hspace{3cm} (7)$$

where ΔH = the expected amount of genetic variation lost in one generation (expressed percent of heterozygosity lost), and Ne is the number of adults in a population, where each member has equal probability of contributing offspring to the next generation (expressed as the effective population size). Based upon this equation the value for Ne

necessary to retain 95% of the genetic variation after a certain number of generations (t = generations) can be expressed as:

$$Ne = 1/[2(1 - 0.95^{(1/t)})] \qquad\qquad (8)$$

To solve the calculation of Ne in Equation 7 for coho, a value for the number of generations (t) expected for a coho population over a period of 100 years (the reference time period) is required. Although coho salmon spawn primarily at age 3, some precocious males (jacks) from each brood spawn at age 2. The proportion of jacks appears to vary among coastal populations, with stream populations averaging about 6% and lake populations averaging about 22% (ODFW, Unpublished data). The result is that average age at maturity for stream populations is 2.94 years (34.01 generations in 100 years). And for lake populations, is 2.78 years (35.97 generations in 100 years). Therefore, substituting these values for t into Equation 8, a Ne of 332 and 351 is obtained for stream and lake populations, respectively.

Developing a Metric for Ne - The calculation of minimum values of Ne for coastal coho are mathematically valid only if coho populations meet the conditions of an "ideal" population. An "ideal" population, whose abundance is represented by the symbol Ne, must meet certain conditions such as: no overlapping generations, equal sex ratio, constant population size, and equal probability of reproductive success for all members of the population. The problem is that nearly all populations (regardless of the species) fail to meet these "ideal" constraints. Therefore, to estimate the rate of genetic loss based upon the number of spawners, a significant number of 'corrections' need to be made. Essentially, the goal of these corrections is to build a metric that represents Ne but which can be applied to the species and data in question.

Typically, when these corrections are made, the number of individuals necessary to achieve the genetic behavior of an "ideal" population is greater than the size of the "ideal" population. By convention, Ne is used to represent the number adults in an ideal population and N the number of adults actually counted in the population. In most real life situations, Ne/N is less than 1.0, (i.e., N is greater than Ne). For salmonids, Allendorf et al. (1997) suggest that Ne/N ratios in the range of 0.20 may be typical. Likewise, McElhaney et al. (2000) suggest that a Ne/N ratio of 0.30 is a reasonable expectation.

One factor that can cause Ne to be less than N is if the probability of producing offspring is unequal among individual spawners. In some extreme examples only a few members of the spawning population produce all of the surviving offspring. In effect, the genetic contribution to the next generation comes from a much smaller number of individuals than a simple count of spawning individuals would suggest. Obviously, the challenge of estimating how many offspring each spawner in a wild population produces is a daunting proposition. However, several estimates of offspring produced per individual spawner have been made for salmonids in recent years (Simon et al 1986, Geiger et al. 1997, and Seamons et al. 2003). Results from these studies can be compared in a standardized manner by calculating V_K/K, where V_K = variance in offspring produced per spawner

and K = mean number of offspring produced per spawner. Among these studies the values for V_K/K are confined to a surprisingly narrow range from 2.63 to 2.96, with a mean of 2.77.

An 'ideal' population represented by Ne is assumed to have a pattern of offspring per parent across all parents that follow a poisson distribution characterized by a ratio of Vk/K ≈ 1. With salmonid populations, apparently having a ratio that is much greater than 1 (for example 2.77 in the studies discussed above), there is a need to adjust Ne calculations to account for this difference. As described by Hartl (2000), the ratio of V_K/K can be used to adjust the calculation of Ne for unequal probabilities of reproductive success using the following equation

$$Ne = 2N/[1 + (V_K/K)] \qquad\qquad (9)$$

When a value of 2.77 (mean of three studies cited above) is substituted for V_K/K in Equation 9, the relationship between N and Ne becomes: Ne = 0.53N. In other words, to account for unequal reproductive success probabilities, N should be reduced by 0.53 in the computation of the effective population size, Ne.

Another factor that needs to be considered in the estimation of Ne is the effect of year-to-year variation in recruitment for species with overlapping generations. For example, Waples (2002) suggests that such variability in Chinook salmon results in a 40% to 60% reduction in the value calculated for Ne. Although compared to Chinook, coho have less variation in age of maturity; Van Doornik et al. (2002) demonstrated that jacks (2-years) contribute enough to gene flow between different age-classes that genetically, coho should be treated as a species with overlapping generations. Therefore, even though the majority of reproduction comes from 3-year old fish, coho should not be treated as pink salmon for which the year classes do not overlap. Therefore, based on these two studies an additional reduction of 60% was applied to the calculation of Ne for coastal coho. Although a 60% reduction is the upper portion of the range suggested by Waples (2002), his analysis was based upon Chinook which have a greater overlap in generations than do coho. Therefore, it seemed appropriate to use the higher reduction value (60%). Indeed, if there was no overlap at all between generations an even higher reduction percentage would have been necessary. The net result of the overlapping generation adjustment and the one for unequal reproductive contribution among families (Equation 9) is the following:

$$Ne = (0.53N)(1\text{-}0.60) = 0.212N \qquad\qquad (10)$$

It is interesting that this level of reduction (Ne = 0.212N) is within the range of values suggested for salmonids by Allendorf et al. (1997) (Ne = 0.20N) and by McElhaney et al. (2000) (Ne = 0.30).

Spawner abundance estimates for salmon are typically collected on an annual basis. However, for a species with overlapping generations (such as salmon) the value for N, such as in Equation 10, needs to incorporate the fact that each year's spawning

population is comprised of individuals from different year classes. Waples (1990b) demonstrated that for Pacific salmon this problem of overlapping generations can be accounted for in the calculation of Ne using the relationship described below.

$$Ne = g \, (Nb) \tag{11}$$

Where g = the average age at maturity for the species and Nb is the harmonic mean of annual counts of spawners for the entire data set. As previously discussed the average age at maturity was g = 2.78 years for lake populations (i.e., Siltcoos, Tahkenitch, and Tenmile) and g = 2.94 years for all other populations (stream populations). The harmonic mean of annual spawner counts (Nb in Equation 11) is calculated from:

$$Nb = 1 \, / \, (1/t)(\sum (1/Ni)) \tag{12}$$

Where t = number of years counted and Ni = number of spawners counted for each year. It should also be noted that using a harmonic mean to characterize spawner abundance essentially gives greater weight to the smaller numbers in the data set. This approach is generally accepted as a way to incorporate the errors that will occur in estimating Ne if population abundance fluctuates widely, as is frequently the case for coho salmon. Finally, bringing together the previous adjustments described by Equation 10 for estimating Ne with those of Equation 11 to account for overlapping generations yields the following equation:

$$Ne = 0.212 \, g \, (Nb) \tag{13}$$

Substituting the average maturity age for lake populations (g = 2.78) and stream population (g = 2.94), Equation 13 can be simplified to:

$$Ne = 0.589Nb \tag{14}$$

For lake populations and

$$Ne = 0.623Nb \tag{15}$$

for stream populations.

As presented earlier (see Equation 8 discussion), Ne values of 351 for lake populations and 332 for stream populations were computed as the Ne that must be achieved to retain at least 95% of the population's heterozygosity over a 100-year time period. Equations 14 and 15 can be used to calculate Ne values directly from an easily measurable quantity – annual spawner abundance, expressed as a harmonic mean over several years. These equations can also be rearranged such that given a desired Ne; the harmonic mean necessary to achieve this Ne can be calculated. As previously stated, the minimum Ne values for coastal coho were 351 and 332 for lake and stream populations, respectively. Translating this to a harmonic mean yields an Nb for lake populations of 351 / 0.589 = 595 and an Nb for stream populations of 322 / 0.623 = 533.

The value for the abundance threshold for the diversity criterion was conservatively established by averaging the critical Nb values for the lake and stream populations (average = 564) and rounding this number upward to 600.

It is possible to use a retrospective or prospective approach to determine if a population meets this Nb abundance threshold. For the purposes of this criterion, only the prospective approach was considered. The output from the population viability model discussed in the persistence criterion section was used to estimate the harmonic mean for a period of time 100 years into the future. Mechanically, a harmonic mean of the spawner escapements was computed for each 100-year trial of the model. After all of the trials were completed (generally there were 3,000 trials per model run), the average of the harmonic means for the individual trials was determined. To 'pass' the diversity criterion the average of the harmonic means had to be greater than 600.

Summary of Population Criteria and Evaluation Thresholds

Population criteria were developed to assess the conservation status of Oregon coastal coho. A summary of each of these and the associated evaluation thresholds are presented in this section of the report.

Criterion 1

Title
Productivity during the worst 7-year period of poor marine survival (Pro1).

Metric
The statistical probability that the mean recruits per spawner was less than replacement (i.e., $R/S > 1.1$) for those years when the spawner density was less than 5 fish per mile.

Evaluation Thresholds
Green - If probability is < 0.25 then the classification is low risk.
Yellow - If probability is > 0.25 but < 0.50 then the classification is a risk warning.
Red - If probability is > 0.50 then the population fails the criterion and is classified at risk.

Criterion 2

Title
Productivity during last 12-year period standardized to the poor marine survival of the early 1990's (Pro2).

Metric
The statistical probability that the mean recruits per spawner was less than replacement (i.e., $R/S < 1.1$) for those years when the spawner density was less than 15 fish per mile and where recruits observed for the 1997 to 2000 brood years were discounted to match

the average marine survival rate experienced by recruits belonging to the 1990 to 1996 brood years.

Evaluation Thresholds
Green - If probability is < 0.25 then the classification is low risk.
Yellow - If probability is > 0.25 but < 0.50 then the classification is a risk warning.
Red - If probability is > 0.50 then population fails the criterion and is classified at risk.

Criterion 3

Title
Forecast probability of extinction within the next 100 years (Per 1).

Metric
The probability of an extinction event occurring as evaluated using a population viability model; where an extinction event is defined as a single occurrence within a 100 year simulation where the 3-year average spawner abundance was less than QET.

Evaluation Thresholds
Green – If probability of extinction is < 0.01 then the classification is low risk
Yellow – If the probability of extinction is > 0.01 but < 0.05, then the classification is a risk warning
Red – If the probability of extinction is > 0.05 then the population fails and is classified at risk.

Criterion 4

Title
Within population distribution of spawners (Dist 1)

Metric
At least 50% of the spawning habitat within a population's home range must be occupied. Occupied is defined as at least 50% of the 6th field HU sub-basins with spawning habitat have a spawner density greater than 4 spawners per mile in at least 50% of the survey sites. Viability criteria for this metric were evaluated for ODFW coho spawner surveys conducted from 1990 to 2000, a period of low coho abundance and poor ocean survival. .

Evaluation Thresholds
Green – Greater than 50% of the 6th field HU sub-basins are occupied by spawners at criteria levels.
Yellow – 40-50% of the 6th field HU sub-basins are occupied – but there is concern that some of the sub-basins included in the analysis do not have sufficient spawning habitat to support spawners at criteria levels.
Red – Less than 50% of the 6th field HU sub-basins are occupied.

Criterion 5

<u>Title</u>
Forecast population abundance in next 100 years is sufficient to prevent the loss of genetic diversity.

<u>Metric</u>
The average of the100-year harmonic means of spawner abundance as forecast from a population viability model is greater than the critical threshold of 600.

<u>Evaluation Thresholds</u>
Green – If 100-year harmonic mean is >1200 then the classification is low risk.
Yellow – If 100-year harmonic mean is less than <1200 but >600 then the classification is a risk warning.
Red – If 100-year harmonic mean is <600 then the population fails and is classified at risk.

VI - Integration of Population Criteria for ESU Evaluation

Most of the discussion to this point has been focused on the evaluation of individual populations. However, to obtain a determination for the entire Oregon coastal coho ESU it was necessary to develop a protocol by which the assessment results for individual populations could be 'rolled up' into a full ESU assessment. The approach for accomplishing this 'roll up' is similar to what has been developed by the OTRT.

For the entire ESU to be classified as not at risk and therefore a candidate for de-listing, all 5 strata must pass a conservation risk threshold. This risk threshold has been defined in terms of population criteria results. To pass the risk threshold a stratum must have either 2 or 50% of the independent populations pass all of the population criteria described earlier. To facilitate this assessment a 'combined' score was given for each population based upon the results of testing its status relative to the five population criteria. A population was classified with a failing combined score, if the population failed any of the criteria. Further, of the populations that pass the criteria, at least one must be classified by the OTRT as "functionally independent". An exception to this requirement is made for the lakes population strata for which all of the populations are potentially independent (i.e., no functionally independent populations exist). Finally, an additional ESU-wide threshold that at least 10 of the independent populations must meet all population criteria, regardless of strata, for the ESU to be classified as 'not at risk'...

VII – Coho Status Relative to Population and ESU Level Criteria

Population Results
Following the methods described, data for Oregon coastal coho populations were analyzed to determine their status relative to the biological criteria. The results were mixed. In general, it appears that populations in the northern portion of the ESU are at

greater risk than those further to the south (Table 4). The failure rates for criteria 2 and 3 are higher for the northerly populations. Also noteworthy is that for Criterion 1 an evaluation was impossible for many of the populations because spawner densities were rarely less than 5 fish per mile. Recall that for Criterion 1, the R/S mean is calculated only for those data points where spawner density was less than 5 fish per mile. Only in the northern portion of the ESU were spawner densities during the poor marine survival years of the mid-1990s observed in the sub-5 fish per mile range. Also note that except for Criterion 4 (distribution), none of the criteria were evaluated for the Floras and Sixes populations. For these two populations it was not possible to calculate the metrics for testing most of the criteria because the data time series was too short. The actual biological condition of these two populations is at this point, essentially unknown.

Table 4. Results of an evaluation of 19 populations of Oregon coho with respect to five biological criteria designed to assess conservation status. Functionally independent populations are listed in **bold** type.

Population	Criterion 1 (Productivity 1)	Criterion 2 (Productivity 2)	Criterion 3 (Persistence)	Criterion 4 (Distribution)	Criterion 5 (Diversity)	Combined
Necanicum	0.113	0.223	0.000	100%	1300	pass
Nehalem	**0.364**	**0.472**	**0.021**	73%	7159	Pass
Tillamook	**0.393**	**0.448**	**0.138**	83%	847	**Fail**
Nestucca	0.144	0.044	**0.029**	85%	1429	Pass
Salmon	--	**0.917**	**0.999**	**0%**	**1**	**Fail**
Siletz	**0.302**	**0.646**	**0.087**	67%	1006	**Fail**
Yaquina	--	0.058	**0.013**	80%	1586	Pass
Beaver	--	--	0.000	100%	2737	Pass
Alsea	--	**0.875**	**0.400**	**50%**	**234**	**Fail**
Siuslaw	--	**0.428**	**0.013**	65%	5999	Pass
Lo Umpqua	--	0.131	0.000	56%	7772	Pass
Up Umpqua	--	**0.976**	0.000	**47%**	10753	**Fail**
Coos	--	--	0.000	73%	12381	Pass
Coquille	--	**0.272**	0.000	60%	9420	Pass
Floras/Sixes	--	--	--	55%	--	--
Siltcoos	--	0.049	0.000	100%	5681	Pass
Tahkenitch	--	0.022	0.000	100%	2318	Pass
Tenmile	--	--	0.000	100%	10706	Pass

A second conclusion from the results is that if a population fails one of the criteria, it is generally likely to fail a second or a third criterion as well. For example, the Alsea population fails criteria 2, 3, and 5. The fact these independent measures of population health give consistent readings, reinforces the strength of the overall population assessment approach.

Finally, the lakes stratum was the only one for which all of the populations passed all of the criteria (at least for those that could be tested). This is consistent with the view that this stratum is probably the most robust and extinction resistant of the 5 strata that comprise the coastal coho ESU.

ESU Results

The biological analysis utilized five criteria to address four viability attributes. In no cases did a population fail all criteria, and in some cases a population failed only one of the criteria. As a precautionary approach to risk assessment, any population that failed one or more of the criteria was designated "at risk" (red designation). As another precautionary approach, the "not at risk" (green designation) category was further divided to identify populations that passed the criteria but only narrowly (yellow designation or "risk warning" category). Consistent with the precautionary approach, a yellow result for any one of the criteria resulted in a "risk warning" designation for the population.

At least 50% of the independent populations in each stratum pass all of the population criteria (Table 4). In addition, each stratum contains at least one 'functionally independent' population that passes all criteria. The exception is the lakes stratum which is entirely comprised of 'potentially independent' populations and therefore exempt from the 'functionally independent' population requirement. Across the ESU, 12 populations pass all criteria, 5 fail (do not pass one or more population criteria), and the status of 2 populations (Floras and Sixes) is unknown. Based upon the conservation criteria developed for this ESU, the results obtained in this evaluation lead to the conclusion that the Oregon coast coho ESU is not at risk and may not warrant listing as a threatened species under the ESA.

VIII – Synthesis and Additional Discussion

Sensitivity to Measurement Error and Analytical Assumptions

Flaws can be found with each of the population criteria used to assess the viability health of Oregon coastal coho. However, the essential question is whether these shortcomings were so large that they have biased the results and yielded an outcome that is inconsistent with the true condition of the ESU with respect to its extinction risk.

Potential sources of error in this evaluation relate to five general topics: 1) basic data measurement error, 2) the span of years with data was shorter than optimal for a reliable evaluation, 3) uncertainty in the future with respect to habitat condition, marine survival, and hatchery programs, 4) validity of assumptions, and 5) potential for selecting criteria that were too high or too low. These topics are discussed for each population viability criteria.

Criteria 1 and 2 – Productivity at Low Spawner Density

As with all of the criteria, the basic data that went into the analyses were spawner abundance estimates. As discussed in Part 2, each year's estimate was derived from an expansion of spawner counts made for index survey sites. Since these sites comprised only a small portion of the total basin, in terms of total stream miles, their expansion to a basin-wide population estimate was a source of error. For any year's population estimate the confidence intervals ranged from the 30% to more than 100%. Therefore, the

differences that were observed between years and any metric based on these observations such as R/S, is an unknown combination of measurement error and true annual variation. Had it been possible to eliminate the measurement error from these data, the standard deviation associated with each data set would have most likely decreased. This would have increased the power of comparing the mean R/S observations to 1.1 as required for both productivity criteria. However, given that the data included variations due to measurement error, to have a low probability that the mean R/S was greater than 1.1 required that the value of the R/S data points had to be higher than would have been the case if the data did not contain measurement error (i.e., less variation and a smaller standard deviation) resulting in a precautionary approach to the productivity analyses.

The other concern was the possibility that the population estimation methodology was consistently over-estimating the true population size. Although more work needs to be done on this question, preliminary information from a study on the Smith River (tributary to the lower Umpqua) suggests that the reverse may be true. Data from an ODFW research study shows that the number of coho spawning above Smith River Falls as estimated by mark-recapture techniques has been consistently larger than the population estimate for the same area based on the same survey methods used elsewhere on the Oregon coast. If these results are representative, it is possible that the present spawner abundance methodology tends to under-estimate rather than over-estimate the true escapement. Obviously, from a conservation assessment standpoint, utilizing a method that tends to under-estimate abundance is more acceptable because it is precautionary.

Another problem with population estimates was allocating the estimate between wild fish and naturally spawning hatchery fish. For example, the percentage of naturally spawning hatchery fish generally averaged less than 10% for most of the populations over the data period from 1990 to 2003 (Table 5). However, for six populations more than 10% of the natural spawners were hatchery fish. For these six populations (Nehalem, Tillamook, Salmon, Siletz, Alsea, and upper Umpqua), errors in estimating the proportion of hatchery fish could have had substantial impacts on the accuracy of the R/S calculations. However, all of these populations failed the population criteria except the Nehalem (Table 5). Therefore, it did not appear that errors in estimating the frequency of hatchery spawners caused a bias towards a positive status determination. Indeed the opposite appeared to be true; all of the populations that failed the population criteria had a percentage of naturally spawning hatchery fish greater than 10%.

Most of the comparisons of the observed R/S values to the threshold value of 1.1 were done with less than 5 points for Criterion 1 and less than 8 points for Criterion 2. In fact, the majority of the populations did not have enough data from spawner densities less than 5 fish per mile to even perform a test for Criterion 1. With so few data points, the likelihood of statistically demonstrating that the R/S data were not drawn from a distribution having a mean R/S equal or less than 1.1 requires either that the variation among data points be slight or that the difference between the observed R/S data points and 1.1 be quite large. As has been discussed, the variation among R/S data points was considerable, partially because of measurement error. Therefore, that populations passed either of these productivity criteria is best explained by the fact that in most cases the

observed R/S values were consistently much greater than 1.1 providing a precautionary approach. Although, the use of additional data points would help with the precision of the evaluation, this was not possible given the currently available data and the retrospective nature of both criteria.

However, one advantage of these retrospective productivity criteria is that they require no assumption about a future set of conditions (freshwater or marine). This is a distinctive advantage over the persistence (C3) and diversity criteria (C5), both of which require some kind assumption about the most likely condition of freshwater habitat and marine survival rates for a period 100 years into the future.

Table 5. Estimated average percentage of hatchery fish in 17 naturally spawning populations Oregon coastal coho, 1990 to 2003.

Population	Average Percent Hatchery Fish	Status[a]
Necanicum	1%	Pass
Nehalem	35%	Pass
Tillamook	16%	**Fail**
Nestucca	1%	Pass
Salmon	61%	**Fail**
Siletz	14%	**Fail**
Yaquina	7%	Pass
Beaver	8%	Pass
Alsea	14%	**Fail**
Siuslaw	7%	Pass
Lo Umpqua	6%	Pass
Up Umpqua	42%	**Fail**
Coos	1%	Pass
Coquille	2%	Pass
Floras/Sixes	1%	--
Siltcoos	0%	Pass
Tahkenitch	0%	Pass
Tenmile	0%	Pass

[a]Combined result from conservation status evaluation using 5 population criteria (see Table 3).

The underlying assumption for both productivity criteria is that if the population is able to replace itself at low spawner densities when marine survival is extremely low, then the population has demonstrated the ability to withstand a severe environmental challenge and this can be interpreted as an indication that the population is a not at risk of extinction. Although this assumption is logical, its weakness is that the data permit only one look in time at such a challenge. It begs the question as to whether one challenge is sufficient to demonstrate that a population will respond positively to the next occurrence of a similar period of poor marine survival rates. Is the proper interpretation of a population surviving one period of poor marine survival that it is healthy, or just lucky? Obviously, with the limited time series of available data this question can not be empirically tested. The problem of how many times can a population be challenged

before it fails is not a question that lends itself to an empirical answer for a long lived species like coho salmon. This type of question is better explored with prospective modeling exercises, such as the one used for the persistence criteria, where repetitive challenges can easily be generated. However, it is worth considering that the results for the productivity criteria, while performed on one population at a time; they were performed for the same environmental challenge. Therefore, if the results across all populations are considered together they could be viewed as pseudo-replicates for the same treatment. From this perspective, the fact that approximately two-thirds of the populations passed the criteria is a useful consideration.

Criteria 3 and 5 – Persistence and Diversity

Both the persistence and diversity criteria were based upon simulations using a modified Ricker recruitment model. The parameters for this recruitment function were estimated via multiple regression for each coho population. In looking at population specific recruitment model parameter estimates and associated statistics it was clear that while all but four of the estimating multiple regressions were statistically significant, it was rare that estimates for both the spawner density parameter (B) and the marine survival parameter (c) were statistically significant (Table 6). Further, in at least one case, the Salmon River population, the estimated value for the marine survival parameter was nonsensical in that it was a negative (-0.17). A negative value implies that as marine survival goes up the productivity of the population goes down.

However, such results are not surprising considering that the parameters estimated for the recruitment function were determined from a multiple regression of only 11 data points. The problems associated with using such a short time series to estimate these parameters are not insignificant. A difficulty perhaps only compounded by the uncertainty in the abundance estimates themselves.

To explore the issue of inaccurate parameter estimation further, a series of sensitivity analyses were performed for each of recruitment parameter (i.e., a, Smax, and c). In these analyses, values for each parameter were either increased or decreased from the base estimate that was obtained for each population (Table 6) and re-run through the viability model to produce new estimates of extinction probability. For example, six proportional values of the 'a' parameter were tested for each population equal to 0.52, 0.64, 0.80, 1.25, 1.55, and 1.93 of the base estimate (Table 7). In other words, applying these proportions to the data for the Necanicum population, for example, yielded test values for the 'a' parameter of 1.9, 2.2, 2.4, 2.6 (base estimate), 2.8, 3.1, and 3.3. Transformed from their natural log form this is equaled to a range in values for intrinsic productivity of 7.1 R/S to 26.6 R/S.

As illustrated in Table 7, the probability of extinction estimates were sensitive to relatively modest reductions in the value for the 'a' parameter (intrinsic productivity) from the value estimated for each population. The results also suggest that over-estimating the 'a' parameter would cause a greater misreading of population extinction

risk, than would under-estimating the 'a' parameter. It was also evident that populations in the northerly portion of the ESU would be more sensitive to an over-estimate error than would be populations in the southern portion of the ESU.

The impact of errors in estimating the habitat capacity parameter, Smax, seemed much less than for the 'a' parameter. The probability of extinction for all populations remained virtually unchanged for a range of 'Smax' parameter values from 0.50 to 3.0 of the base value (Table 8).

For the third equation parameter related to marine survival, 'c', the pattern of sensitivity appeared nearly a mirror image of what was observed for the 'a' parameter. In the case of the 'c' parameter increases beyond the base values resulted in a sharp increase in the extinction probability (Table 9). Therefore, underestimates with respect to the 'c' parameter (i.e., the true parameter is greater than the one estimated) could lead to a under prediction of the true extinction risk. It is also notable, that as with the 'a' parameter estimates, the potential for modest parameter errors causing substantial problems with the extinction risk predictions is higher for populations in the northern portion of the ESU.

Table 6. Ricker recruitment equation parameters **a**, **B**, and **c** and associated statistics estimated for 19 populations of Oregon coastal coho from population abundance data collected from 1990 to 2003. With the exception of the Criterion 3 results, all shaded cells refer to results that were not statistically significant ($P > 0.05$).

Population	**a** Intrinsic Productivity	**B** Expressed as Smax	**c** Marine Survival	Standard Deviation of Regression	R^2	Result for Criterion 3 (Persistence)
Necanicum	2.620	587	1.00	0.766	0.558	0.000
Nehalem	2.447	7229	1.50	0.576	0.847	**0.021**
Tillamook	2.224	2622	1.21	0.915	0.538	**0.138**
Nestucca	2.833	1505	1.34	1.010	0.501	**0.029**
Salmon	0.446	633	-0.17	0.834	0.362	**0.999**
Siletz	1.750	2729	0.98	0.723	0.633	**0.087**
Yaquina	2.586	1692	0.56	1.393	0.433	**0.013**
Beaver	1.721	2465	0.68	0.713	0.192	0.000
Alsea	1.430	5161	0.96	1.074	0.383	**0.400**
Siuslaw	3.189	2906	1.08	1.011	0.630	**0.013**
Lo Umpqua	2.464	3215	0.51	0.879	0.458	0.000
Up Umpqua	1.750	8319	0.81	0.467	0.518	0.000
Coos	2.614	4521	0.19	0.994	0.652	0.000
Coquille	1.899	5831	0.46	0.642	0.578	0.000
Floras/Sixes	--	--	--	--	--	--
Siltcoos	2.491	1676	0.23	0.556	0.613	0.000
Tahkenitch	1.639	1812	0.24	0.716	0.180	0.000
Tenmile	2.449	3216	0.42	0.402	0.593	0.000

Table 7. Sensitivity of probability of extinction forecasts to changes in the estimated value for the 'a' parameter (intrinsic productivity) of the recruitment function for 17 populations coastal coho.

	Proportional Change in Ricker a Value from Base						
	0.52	0.64	0.80	Base	1.25	1.55	1.93
Necanicum	0.12	0.02	0.00	0.00	0.00	0.00	0.00
Nehalem	0.91	0.61	0.23	0.01	0.00	0.00	0.00
Tillamook	0.83	0.63	0.37	0.15	0.03	0.00	0.00
Nestucca	0.49	0.24	0.10	0.03	0.00	0.00	0.00
Salmon	1.00	1.00	1.00	1.00	0.94	0.52	0.12
Siletz	0.93	0.73	0.35	0.08	0.01	0.00	0.00
Yaquina	0.30	0.17	0.05	0.01	0.01	0.00	0.00
Beaver	0.01	0.00	0.00	0.00	0.00	0.00	0.00
Alsea	0.98	0.90	0.67	0.38	0.19	0.04	0.01
Siuslaw	0.35	0.19	0.06	0.02	0.00	0.00	0.00
Siltcoos	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tahkenitch	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Lower Umpqua	0.01	0.00	0.00	0.00	0.00	0.00	0.00
Upper Umpqua	0.66	0.14	0.01	0.00	0.00	0.00	0.00
Tenmile	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coos	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coquille	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 8. Sensitivity of probability of extinction forecasts to changes in the estimated value for the 'Smax' parameter (habitat capacity) of the recruitment function for 17 populations coastal coho.

	Proportional Change in Smax Value from Base						
	0.50	0.70	0.85	Base	1.50	2.00	3.00
Necanicum	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nehalem	0.03	0.03	0.03	0.02	0.03	0.03	0.03
Tillamook	0.19	0.18	0.16	0.15	0.16	0.16	0.16
Nestucca	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Salmon	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Siletz	0.11	0.10	0.10	0.09	0.10	0.10	0.10
Yaquina	0.01	0.02	0.02	0.02	0.02	0.02	0.02
Beaver	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Alsea	0.41	0.39	0.38	0.37	0.38	0.39	0.42
Siuslaw	0.01	0.02	0.01	0.01	0.02	0.02	0.02
Siltcoos	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tahkenitch	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Lo Umpqua	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Up Umpqua	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tenmile	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coos	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coquille	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 9. Sensitivity of probability of extinction forecasts to changes in the estimated value for the 'c' parameter (marine survival) of the recruitment function for 17 populations coastal coho.

	Proportional Change in 'c' Parameter Value from Base						
	0.52	0.64	0.80	Base	1.25	1.55	1.93
Necanicum	0.00	0.00	0.00	0.00	0.01	0.13	0.60
Nehalem	0.00	0.00	0.00	0.02	0.40	0.93	1.00
Tillamook	0.00	0.00	0.02	0.15	0.53	0.83	0.98
Nestucca	0.00	0.00	0.00	0.03	0.18	0.50	0.82
Salmon	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Siletz	0.00	0.00	0.00	0.09	0.56	0.92	1.00
Yaquina	0.00	0.00	0.00	0.02	0.05	0.10	0.27
Beaver	0.00	0.00	0.00	0.00	0.00	0.00	0.10
Alsea	0.00	0.02	0.13	0.39	0.66	0.93	1.00
Siuslaw	0.00	0.00	0.00	0.02	0.10	0.35	0.72
Siltcoos	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tahkenitch	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Lower Umpqua	0.00	0.00	0.00	0.00	0.00	0.01	0.06
Upper Umpqua	0.00	0.00	0.00	0.00	0.03	0.66	0.99
Tenmile	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coos	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coquille	0.00	0.00	0.00	0.00	0.00	0.00	0.00

In summary, the basic finding of the sensitivity analysis was that extinction risk predictions were not sensitive to errors in the estimation of the 'Smax' parameter; however they were sensitive to overestimates of the 'a' parameter and underestimates of the 'c' parameter. The primary source of potential error in the parameter estimates is probably the fact that so few points (n = 11) were used in the estimation process via multiple regression analysis. However, as described earlier and illustrated in Figures 8, 9, and 10, a comparison of parameter estimates generated from a long-term (n = 46) versus short-term (n = 11) analysis of data from the same spawning survey sites did not provide evidence of an overall positive or negative bias with respect to the values obtained for the 'a' and 'c' parameters. In other words, it did not appear that reliance on the short-term data set to estimate these two parameters resulted in a consistent pattern of over or under parameter estimation (assuming the long-term parameter estimates were the most accurate of the two time series). In contrast, it did appear that the values for the 'Smax' parameter from the shorter time series were less than for the longer time series. Although this could be interpreted as the shorter time series having an underestimation problem, the sensitivity analysis suggests that such errors would have little impact on the extinction risk predictions from the viability model. Therefore, extinction predictions are most effected by errors in parameters that do not appear to be overly sensitive to the problem of using a shorter time period for their estimation ('a' and 'c'). Further the estimation of the 'Smax' parameter, while sensitive to the time period used, is of less concern because variations in the value for 'Smax' have so little impact on the extinction risk forecasts.

Assumptions about Future Freshwater and Marine Conditions - The model runs performed to estimate the probability of extinction for each of the 17 populations assumed that the pattern of marine survival rates experienced by the 1958-2000 broods were predictive of marine survival for the future 100 year period of the model. Likewise, the conditions of the freshwater habitat during the population estimate time period (1990 to 2003) were assumed to representative of the condition of the habitat in future. However, one or both of these assumptions could be wrong. To examine the implications of habitat and survival scenarios for the future different than the ones modeled, additional analyses using the viability model were performed. These "what if" analyses were structured to look at two primary factors: life cycle survival and habitat capacity. Although the habitat capacity is directed at the freshwater portion of the life history and directly related to the amount of space a population has available to live in, there are multiple factors that can impact life cycle survival including: habitat quality, marine conditions, fishing mortality, and interactions with hatchery fish.

The approach to these "what if" exercises were analytically similar to the methods previously described for the sensitivity analyses. However, rather than a focus on parameter estimation errors, the central question was how would the probability of extinction change if survival rates or habitat capacities were different in the future than they had been in the past.

The first set of analyses was directed at future differences in life cycle survival. A number of different "what if" scenarios were performed with the assumed life survival ranging from 1/10 to 3 times the survival rate that has been observed over the last 46 years. As presented in Table 10 and illustrated in Figure 11, modest reductions in life cycle survival from the current status (last 43 years) resulted in a sharp increase in the probability of extinction for populations in the northern portion of the ESU. However, an order of magnitude decline in survival rate put all of the populations in the ESU at significant extinction risk. In contrast, increasing the current survival rate by a factor of 2.5 resulted in all populations that are at risk under current conditions to 'climb out' of this risk zone.

It is emphasized that regardless of the stage in the life history, a decrease in survival is carried throughout the life cycle. For example, a 15% reduction in egg to emergent fry survival across an entire population would have the same impact as a 15% reduction in over-winter survival of juveniles, or a 15% reduction in marine survival rate, or a 15% increase in the mortality impact of fisheries. In addition, because survival is multiplicative, it is possible to use the results in Table 10 to gauge the net effect of changes in survival at several life stages at once. For example, a 30% decrease in freshwater survival due to habitat modifications on top of a 30% decrease in marine survival would result in an overall decrease in life cycle survival of $[1 - (1 - 0.30)(1 - 0.30)] = 0.51 * 100\% = 51\%$. The multiplicative feature of the survival relationship can be used to examine any number of possible combinations of changes from the current assumption about the future of survival through certain portions of the coho's life history. However, it should be noted that the way the persistence model modeled these changes was as if they occurred immediately at the beginning of the 100 year test period.

Therefore, if the survival changes occur gradually as a slow decline or increase, the model will likely overestimate the risk during the 100 year period over which changes are occurring.

Table 10. Sensitivity of probability of extinction forecasts to changes in life cycle survival for 17 populations coastal coho.

|	Survival Rate Relative to Current										
	0.10	0.20	0.50	0.70	0.85	Current	1.20	1.45	2.00	2.50	3.00
Necanicum	1.00	1.00	0.14	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nehalem	1.00	1.00	0.93	0.43	0.17	0.02	0.00	0.00	0.00	0.00	0.00
Tillamook	1.00	1.00	0.87	0.55	0.33	0.14	0.06	0.01	0.00	0.00	0.00
Nestucca	1.00	1.00	0.60	0.18	0.07	0.03	0.01	0.00	0.00	0.00	0.00
Salmon	1.00	1.00	1.00	1.00	1.00	1.00	0.96	0.72	0.10	0.01	0.00
Siletz	1.00	1.00	0.97	0.57	0.28	0.09	0.01	0.00	0.00	0.00	0.00
Yaquina	1.00	0.99	0.26	0.08	0.03	0.01	0.00	0.00	0.00	0.00	0.00
Beaver	1.00	1.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Alsea	1.00	1.00	0.99	0.85	0.62	0.40	0.20	0.08	0.01	0.00	0.00
Siuslaw	1.00	1.00	0.40	0.19	0.04	0.01	0.00	0.00	0.00	0.00	0.00
Siltcoos	0.97	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tahkenitch	1.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Lo Umpqua	1.00	0.96	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Up Umpqua	1.00	1.00	0.78	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tenmile	0.94	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coos	1.00	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coquille	1.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
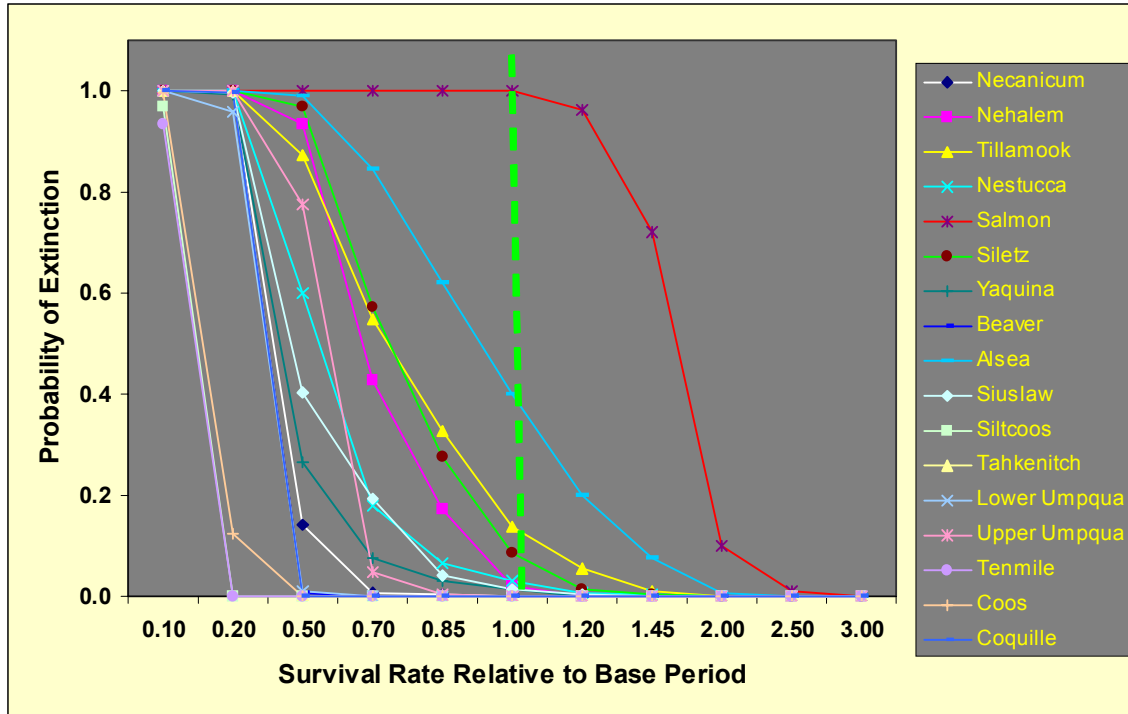
Figure 11. Relationships between probability of extinction and change in life cycle survival for 17 modeled populations.

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The other type of "what if" scenarios explored with respect to projected extinction risk were those that involved changes in the amount of habitat available to each population. The results from these scenarios were markedly different from those obtained when focus was life cycle survival. First, it appears that increasing the capacity of habitat for populations at risk under current conditions will not result in a reduction of this risk (Table 11 and Figure 12). Second, for those populations that are not at risk under current conditions it took a considerable reduction in habitat capacity (50% to 80%) before they fell into the at risk category. Although this result may be somewhat surprising, it in fact demonstrates that with respect to extinction risk these populations are much more sensitive to changes in the quality of their habitat (survival related) than they are to changes in the quantity of the their habitat (capacity related). Therefore, the assumptions about future habitat capacity are not as important to the accuracy of extinction risk predictions, especially considering the improbability of eliminating 50% to 80% of the remaining coho habitat in the near future – as this is the range of change necessary to elicit substantial changes in the risk of extinction. However, as a caveat, it should be stated that if habitat that is lost under a "what if" scenario is disproportionately of high quality then the analysis should be redirected to look at the associated changes to life cycle survival.

Table 11. Sensitivity of probability of extinction forecasts to changes in habitat capacity for 17 populations coastal coho.

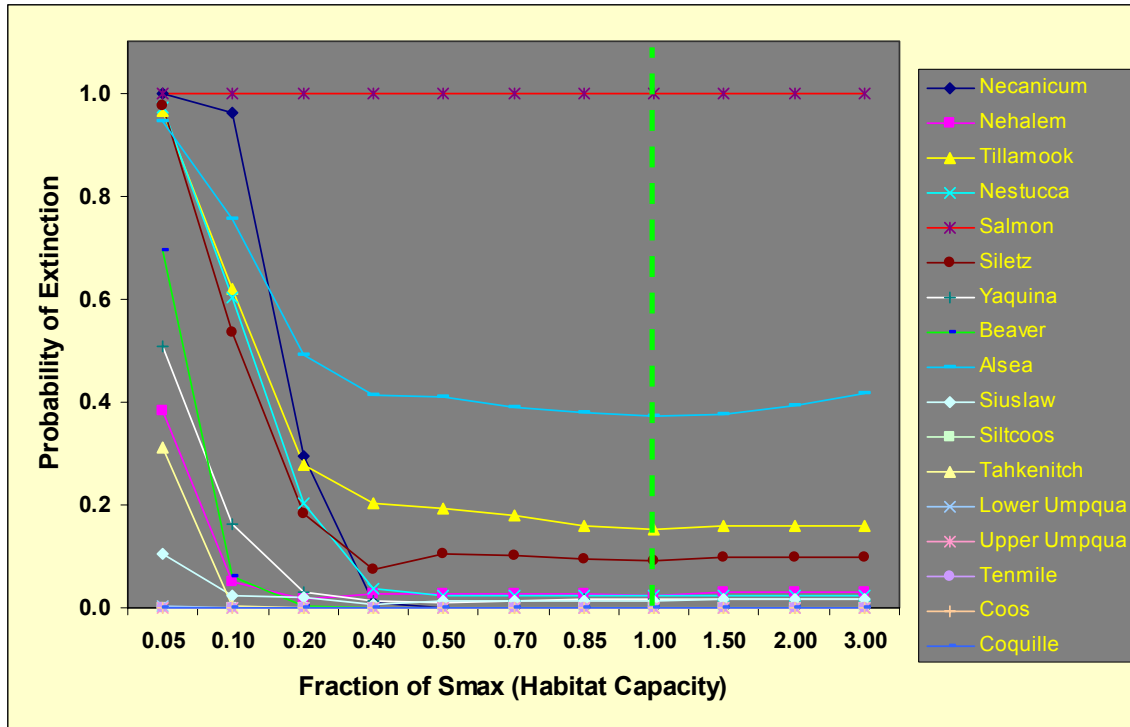|	Habitat Capacity Relative to Current										
	0.10	0.20	0.50	0.70	0.85	Current	1.20	1.45	2.00	2.50	3.00
Necanicum	1.00	0.96	0.30	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nehalem	0.38	0.05	0.02	0.03	0.03	0.03	0.03	0.02	0.03	0.03	0.03
Tillamook	0.96	0.62	0.28	0.20	0.19	0.18	0.16	0.15	0.02	0.16	0.16
Nestucca	0.97	0.60	0.20	0.04	0.02	0.02	0.02	0.01	0.02	0.01	0.02
Salmon	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Siletz	0.97	0.53	0.18	0.07	0.11	0.10	0.10	0.09	0.10	0.10	0.10
Yaquina	0.51	0.16	0.03	0.01	0.01	0.02	0.02	0.02	0.01	0.01	0.02
Beaver	0.69	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Alsea	0.94	0.75	0.49	0.41	0.41	0.39	0.38	0.37	0.38	0.39	0.42
Siuslaw	0.11	0.02	0.02	0.01	0.01	0.02	0.01	0.01	0.02	0.02	0.02
Siltcoos	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tahkenitch	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Lo Umpqua	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Up Umpqua	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tenmile	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coos	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Coquille	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Figure 12. Relationships between probability of extinction and change in habitat capacity for 17 modeled populations.

Impact of Hatchery Programs - As has been previously discussed there appears to be an association between populations classified at risk and the proportion of hatchery fish in the natural spawning population that was greater than the ESU-wide average. Although it is not certain if these hatchery programs are one of the main problems for these populations, the possibility exists. Nickelson (2003) has suggested that the addition of hatchery reared coho smolts to a basin has a negative influence on the survival of wild coho smolts via the attraction of predators. Given that there were substantial reductions in hatchery smolt release numbers during the late 1990s for many of the populations classified as at risk, it possible that in the near future the effective survival rate for these populations could improve. In addition, the likelihood that naturally spawning hatchery fish are less successful in producing offspring than wild fish (Leider et al. 1990; Reisenbichler and Rubin 1999; Kostow et al. 2003; and McLean et al. 2003) reducing the number of naturally spawning hatchery fish within a population would improve the conversion of spawners to recruits and thereby increase overall population productivity. Again, it will be several more years before the benefits of these reductions in hatchery programs are known, however it is possible the net effect may be to increase life history survival enough to lift several of the populations out of their currently at risk classification. For example, considering the reduction in the number of hatchery smolts released, the expected improvement life history survival – based on the relationship described by Nickelson (20003) – would be an increase of 40%, 38%, and 110% for the Nehalem, Siletz, and Alsea populations, respectively. If realized this would be a sufficient to reduce the probability of extinction (see Table 9) and remove the current "at risk" classification for these populations. That the 'cure' for these at risk populations lies

in the improvement of some factor that improves life cycle survival and not habitat capacity, as previously discussed, adds more support to the hypothesis that hatchery fish may be the primary problem for these populations.

Diversity Criterion – The harmonic mean of spawner abundance that was the metric of the diversity criterion was tied to the output from the persistence model. Therefore, the population results for this criterion were subject to same influence of errors in estimation of the recruitment equation parameters as the persistence criterion. The results were also susceptible to which assumption was used concerning the condition of the freshwater and marine environment in the future. However, there were some differences between the response of this criterion and the persistence criterion to lowering or raising the expectations for survival in the future. First, the number populations failing the metric for this criterion were less than for the persistence criterion. However, as the assumed future survival rate was decreased, it appeared the transition from a few populations failing the criterion to nearly all failing was sharper (Table 12). It also appeared the north to south pattern of sensitivity to decreases in survival rate was less variable. However, in general the probability of extinction and the harmonic mean responded in a similar fashion to decreases in assumed survival rates.

Table 12. Results for diversity criterion for 17 populations of Oregon coast coho (see text).

	Survival Rate Relative to Current										
	0.10	0.20	0.50	0.70	0.85	Current	1.20	1.45	2.00	2.50	3.00
Necanicum	1	10	209	524	875	1247	1622	2062	2503	3107	3704
Nehalem	1	2	11	212	1517	4783	8717	12685	16817	21442	26720
Tillamook	1	1	9	46	277	730	1905	3184	4636	6362	8453
Nestucca	1	3	119	346	757	1517	2166	3323	4364	5263	6579
Salmon	1	1	1	1	1	1	2	23	136	396	614
Siletz	1	1	1	29	274	1012	2184	3460	4917	6307	8071
Yaquina	2	10	248	516	941	1595	2446	3248	4405	5535	6891
Beaver	2	23	712	1428	2087	2745	3622	4621	5811	7303	8978
Alsea	1	1	2	5	64	160	766	1819	3570	5215	7490
Siuslaw	4	36	801	1769	3505	6133	8872	11492	14934	17810	20509
Siltcoos	2	945	2877	3584	4456	5549	6812	8466	10320	12578	15074
Tahkenitch	2	95	961	1377	1814	2347	2978	3718	4626	5725	7091
Lo Umpqua	4	335	2729	4147	5593	7253	9166	11472	14094	17031	20453
Up Umpqua	1	2	206	1991	6028	9994	13356	17077	21697	27115	33705
Tenmile	4	1712	5588	7058	8767	10861	13336	16475	20271	24771	29928
Coos	2	542	5093	6713	8390	10600	13044	16170	19903	24038	29117
Coquille	1	30	4182	5829	7678	9630	12225	15152	18765	23175	28620

Risk of Catastrophic Events - Risk of natural stochastic events exerting additional risk to population viability at low abundance was addressed in the persistence analysis, which incorporated natural and sample variability evident over the past 50 years. Risk of catastrophic events exerting additional risk to population viability was also considered in the persistence modeling. The assumption was that the greatest catastrophic risk to the coastal coho ESU was actually a significant downturn in marine survival associated with unfavorable ocean conditions. Rather than being hypothetical, this event occurred and was analyzed explicitly. Other potential causes of catastrophic risk were considered and

assumed minor at the ESU level (e.g., fire, flood, drought, tsunami, etc.). Risk at population levels is obviously higher, but the probability of these events impacting populations throughout the ESU is assumed low.

<p align="center">**Criterion 4 – Distribution**</p>

Unlike the persistence and diversity criteria, the distribution criterion was based on a retrospective metric, so it was not complicated by assumptions about predictive models and errors in the estimation of recruitment equation parameters. The problem of errors associated with the population abundance estimates used for the productivity criterion were also avoided because the metric for the distribution criteria was the direct count of fish observed in the selected survey sections. Consequently, the distribution criterion was the least impacted by potential errors of estimation or measurement of all the criteria used in this evaluation. However, less clear was the precise biological meaning of any the data collected to evaluate this metric. For example, an R/S value less than one for a population under the circumstance of low spawner density has clear negative implications for the future of the population. However, the biological consequences of less than 50% of a population's 6th field HUs being occupied in any given year are less obvious. Therefore, the selection of a 50% threshold is not a precisely determined value, other than somewhere in a continuum between 10% occupancy and 90% occupancy it is reasonable to expect a threshold of viability exists.

It is also not clear that the 6th field HUs provide the best means to stratify a basin to quantity the desired interaction between fish and local variations in the habitat. It is possible a more ecological based stratification scheme may be more appropriate. However, from the standpoint of connectedness of sub-units of a fish population within a watershed the ecological overlay also has some shortcomings. Perhaps the best alternative would be some type of double overlay of both watershed and ecotype stratification schemes. However, for the present evaluation it was felt that the geographical approach used provided the most useful alternate for assessing spawner distribution.

IX - Summary of Evaluation Results

This evaluation benefited greatly from the unusual circumstance of an intensive monitoring program functioning across the entire range of the ESU during a period of adverse environmental conditions that were so extreme and unprecedented that they tested the very survival of the species. The wealth of data that recorded the performance of these coho populations during this test period and their rebound when marine survival conditions moderated in the early 2000s formed the core of this conservation assessment.

Multiple, mostly independent criteria were used to evaluate the health of populations within this ESU. The productivity and distribution criteria were based upon a

retrospective look at observed population performance in the recent past. The persistence and diversity criteria were based on a prospective forecast of future population abundance based upon an understanding of the dynamics of coho recruitment gained from both the recent and more distant past. In general, the results from application of these criteria to all 17 independent populations belonging to the ESU produced consistent results. For example, all criteria painted the same picture of the populations in the northerly portion of the ESU being more vulnerable than those in the south and further, that the lake populations (Siltcoos, Tahkenitch, and Tenmile) were likely the most robust populations in the entire ESU.

The application of the assessment protocol for individual populations and collectively the ESU yielded results that were consistent with a "not at risk" classification. However, this classification was based upon several key assumptions concerning the future condition of the marine and freshwater environment. Because of this, the evaluation also included a series of "what if" scenarios to examine the impact of long-term changes in life cycle survival and habitat capacity with respect to the future status of the populations. From these analyses it was apparent that moderate and permanent declines in the condition of the marine and freshwater habitats could put populations at greater risk. However, several of the most vulnerable populations to this additional stress were also the ones that were expected to benefit from major reductions in the presence of hatchery coho and the associated improvement in life cycle survival. Therefore, it is possible that at the ESU level the adverse impact of moderate declines in the marine and freshwater habitat could be offset by the expected benefit of changes in hatchery programs that impact the most vulnerable populations in the ESU.

If even greater declines in the marine and freshwater environments occur that reduce the net life history survival by more than 50% of what was experienced over the last 50 years, then multiple population extinctions in the north and middle portion of the ESU and significant depression of the abundance of the remaining populations could occur. However, changes of this magnitude are unlikely to occur with such rapidity as to preclude a new assessment, status recommendation, and appropriate management response.

It is also noteworthy that although the biological analysis results are sensitive to a declining overall trend in marine survival, the risk associated with this possible trend is not high in the near term. This is because there is an observed decadal oscillation about the mean or trend (Figure 2b) that indicates a high likelihood of generally favorable marine survival conditions for another decade or so. Therefore, based upon the analysis presented in this evaluation and plausible assumptions about the future condition of both the freshwater and marine environments, it is concluded that at this time, the persistence of Oregon coastal coho ESU is not at risk.

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